PHD

Integrated water resources and asset management at a catchment scale: a life-cycle improvement approach

Papacharalampou, Chrysoula

Award date:
2017

Awarding institution:
University of Bath

Link to publication

Integrated Water Resources and Asset Management at a catchment scale: a life-cycle improvement approach

Chrysoula Papacharalampou

A thesis submitted for the degree of Doctor of Philosophy

University of Bath

Department of Mechanical Engineering

February 2017

Acknowledgments

My doctoral research project has been a truly interesting and memorable experience. During this unique journey, I have met and worked with talented people, made new friends and received invaluable support from beloved ones: family and friends. I, thus, take the opportunity to thank all those you have accompanied me in this adventure.

First and foremost, I would like to thank my supervisors, Dr Marcelle McManus and Prof Linda Newnes for the opportunity they gave me to conduct doctoral research under their supervision and for their support throughout the process. I am particularly thankful to them for believing in me even at times that I had lost faith in myself. They have helped me to develop into an independent and creative researcher; as I have always dreamed.

I would also like to thank the industrial partner and funder of my doctoral project, Wessex Water Services Ltd, for their financial and practical support over the years. Special thanks go to Mr David Peacock, Dr Dan Green, Mr John Hayes and Mr Jon Wright for facilitating the smooth progress of the project and providing help at critical stages.

Conducting research can, at times, be quite lonely. I wouldn't be able to make it through without the moral support, love and affection of my family and of a few close friends. My gratitude goes to my parents, Chirstos and Sotiria, for their moral support throughout this journey. Their unconditional love and their faith in me has given me the confidence and the strength to overcome stressful periods. I am equally thankful to my brother, Kostas, whose infinite positive energy has been invaluable during my adventure. Spending time and sharing thoughts with old and new friends has acted therapeutically and helped me maintain a reasonable level of social life interactions. Thanks to all of you: Dimitra Salmanidou, Katerina Michailidou, Evangelia Mitakou, Inge Muhlig, Susan Lattanzio, David Peacock, Chen Ying, George Kolios-Tsiroyannis.

My very special thanks go to my very own, Mimis (or, should I say Dr Dimitris Kallergis?). I am grateful to you for always being by my side, despite the distance or the time difference. This journey would not have been the same without our conversations and your monumental patience to hear me talking about all my jigsaw puzzles. These few words can only summarise my gratitude - thank you from the bottom of my heart.

Abstract

In the water utility sector, traditional asset management focusses on the maintenance and provision of physical assets (infrastructure) that allow water companies to deliver their services, meet their customers' expectations and achieve their economic objectives. Nevertheless, the serviceability of the sector heavily depends on natural elements (e.g. rain, land).

The importance of Natural Capital (i.e. the natural systems and their deriving ecosystem services) has been at the core of policy recommendations which have shaped regulatory changes in the water sector of England and Wales. Water companies are now required to explicitly account for and report their inter-dependencies on the natural environment and adopt systems-oriented approaches in their Asset Management Programmes (AMPs). These reforms will enable the sector to become resilient to the environmental and societal challenges faced at urban and rural contexts.

Responding to the regulatory demands, the research introduces a novel and structured approach for integrating natural capital in the asset management portfolio of the water industry. The work is built on a transdisciplinary research framework and demonstrates that a new scale needs to be considered for the implementation of Holistic Asset Management: the water basin or catchment.

A Catchment Metabolism modelling schema was created, grounded on the principles of Integrated Catchment Management and ecosystems services. The schema is based on the robust synthesis of concepts, tools and methods from a spectrum of disciplines. These include Industrial Ecology, Water Accounting, Environmental Regional Input-Output Analysis, hydrology, software engineering and functional modelling.

Catchment Metabolism introduces a holistic perspective in asset management and expands its scope. The schema enables the conceptualisation, modelling and management of catchments as complex asset systems. It, thus, forms the ground for structured collaboration among experts for integrated water resources planning and decision-making. The schema allows for the design and implementation of catchment-based strategies and the assessment of their environmental performance. An industrial case study for a pilot catchment system (Poole Harbour Catchment) is used to demonstrate the application of the Catchment Metabolism. Alternative strategies for nitrogen pollution mitigation are assessed. The application of winter cover crops across the catchment appears to be the optimum strategy.

The case study demonstrates the practical and modular implementation of the schema, reveals its methodological strengths and limitations and evaluates its applicability in the asset management planning and decision-making of the water sector.

Table of Contents

List of Figures

List of Tables

List of Publications

The following publications have been made to date, as outcomes of the research undertaken:

- Papacharalampou C., McManus M., Newnes L.B., Green D. (2016). Catchment metabolism: Integrating natural capital in the asset management portfolio of the water sector, Journal of Cleaner Production, doi: 10.1016/j.jclepro.2016.11.084.
- Papacharalampou C., McManus M., Newnes L.B., Green D. (2016). Structuring Integrated Asset Management in the UK water sector. 17th IWA Young Water Professionals Conference, 29 March-1 April, Norwich, UK. *In press*
- Papacharalampou C., McManus M., Newnes L.B., Hayes J., Wright J. (2015). Thinking Catchments: a holistic approach to asset management in the water sector. Transitions to the Urban Water Services of Tomorrow (TRUST): IWA Cities of the Future Conference, 28-30 April 2015, Mülheim an der Ruhr, Germany. Proceedings of the Cities of the Future Conference, pp 184-198.

Chapter 1: Introduction

1. Background

Water is one of the most important substances on our planet and of essence for the preservation of life and human well-being, while playing a central role to economic growth. Yet, water is a rather scarce natural resource, irregularly distributed in space and time and exposed to numerous pressures in terms of its quantitative and qualitative status. Climate change, population growth, urbanisation, intensification of agriculture, deforestation and pollution place major threats on water, while the slow travel times of the perpetual natural and virtual water cycles may result in intergenerational disputes and urges the need for long-term strategies. Integrated Water Resources Management (IWRM) plays an important role in balancing out the sustainable exploitation and allocation of the available resources while meeting the ever increasing water demand.

The United Nations World Water Development Report of the World Water Assessment Programme (UN, WWAP 2012) divides water resources management intro three broad activity categories: managing the resource, managing water services and managing the trade-offs needed to balance supply and demand. The complexity of water management, combined with a rapidly changing natural environment and the uncertain socio-economic context, urges the design and implementation of smart strategies through effective actions. The concept and principles of IWRM are highly important to address the complexity; the challenge is to establish an effective approach to implementation with emphasis on the delivery of adequate services (Rouse 2013).

In England and Wales, the Water Act 1973 established a vertical, fully integrated model in regards to water supply and wastewater management. This includes all aspects of these main services: water resources planning, water and wastewater treatment and distribution and customer service. Ever since the privatisation of the industry in 1989, ten water and sewerage limited companies and thirteen water-only companies serve England and Wales. Water and sewerage companies have been effective in delivering significant improvement to their customer service, whilst meeting environmental and regulatory obligations. The achievements of the sector are based on significant investments on asset (infrastructure) projects. According to OFWAT (2006), in the period 1989-2004, water industry invested over £50 billion on capital, while their capital investment programme reaches £16.8 billion on an annual basis, in order to deliver service improvements and maintain existing asset systems. Thus, in the UK water utility sector, the provision and maintenance of physical assets (infrastructure) has, to date, been the focal point of asset management strategies and planning, as a successful mechanism to meet their customers' expectations and achieve their economic objectives.

Nevertheless, the serviceability of the commodity industries, including those of the water sector, heavily depends on the provision of natural elements or assets (e.g. rain, land), as they may have a major impact on them either directly or through their supply chains. The poor management of the Natural Capital (i.e. the world's natural systems and their deriving services) has been related to catastrophic consequences on ecosystems productivity, human wellbeing and financial resiliency (Natural Capital Initiative 2015). In these grounds, the Natural Capital Declaration (NCD, UNEP 2012) demonstrated the willingness and commitment of financial institutions of the private and commodity sectors to integrate Earth's natural assets in their reporting, accounting and decision-making. A considerable number of business initiatives have emerged since, aiming at the integration of natural capital in financial decision-making with special focus on awareness raising, business encouragement and publications (Maxwell et al. 2014).

An essential action requested under the NCD is for companies to disclose the nature of their dependence and impact on Natural Capital through transparent qualitative and quantitative reporting. Several policy initiatives (e.g. UN System for Environmental-Economic Accounting, SEEA) and programs (e.g. World Bank Wealth Accounting and Valuation of Ecosystem Services, WAVES, https://www.wavespartnership.org/) provide a basis for resources accounting by raising the relevance between environmental and financial accounting, but mainly focusing on the economic valuation of natural capital and ecosystem services. Yet, limited work has been done in regards to the development of a standardised approach for the integration of the accounting methods into systems modelling that would allow for the reporting and accounting of the mutual relationships among built, financial and natural assets.

The development of such methodologies would prove particularly important for the water sector, especially in response to policy demands. In the past years, there has been a drive by regulators across Europe to improve the quality of the aquatic environment, in response to the Water Framework Directive (WFD, EC 2000/60). The ventures towards achieving the WFD goals created a growing case for studying and understanding the dependencies that water industries have on natural assets, risks and opportunities associated with this relationship and their real value. In order to adapt to current challenges, the UK water sector is officially encouraged to become more resilient by reforming their asset management practices, adopting integrated approaches and achieving balance between financial costs and environmental impacts (Defra 2016; OFWAT 2015a; OFWAT 2015b; UKWIR 2014).

This research contributes to the expansion of the scope of asset management in the water sector, by introducing a novel methodological approach and modelling schema. The underpinning, transdisciplinary work demonstrates that a new scale for the implementation of asset management needs to be considered: the water basin or catchment. Within these natural, spatial boundaries water companies can study and report on the interdependent relations and impacts of the built capital (physical assets) on the natural environment.

Drawing from literature from a spectrum of disciplines, the modelling schema allows for the design and implementation of asset management strategies at a catchment scale and the assessment of their environmental impacts. In order to evaluate the approach, the research is undertaken for a pilot catchment in collaboration with Wessex Water Services Ltd. The application of the methodology developed for an example catchment justifies whether land management approaches are more efficient, in terms of their environmental performance, in achieving desired water quality status for catchment systems. The research highlights new investment paths and facilitates communication among water companies and their regulators and external stakeholders. Its outcomes will be embodied in the strategic plan of the industrial partner for the next asset management programme (AMP7, 2020-2025) and price review (PR19).

The research provides a novel, comprehensive and structured methodology for holistic and resilient asset management planning in the water sector. It allows the integration of natural capital in the asset management portfolio of the water industry and expands the scope of asset management so that the catchments are modelled and managed as asset systems. In detail, the undertaken research:

- Introduces the catchment as a unit of analysis for asset management purposes.
- Presents a modelling schema which allows to model the catchment as a complex, hybrid asset system. The schema is formulated on a structured and transparent basis, which enables its reproducibility for multiple systems and facilitates communication among experts and stakeholders.
- Builds a systems model for flow accounting at a catchment scale. The model maps the water regime and demand for different actors of the catchment. Mass balance equations and indexes from water and environmental accounting form its mathematical structure. The model can serve as a structure to map and account for other environmental and economic flows within the catchment boundaries.

- Addresses and tests the use of indexes common in supply chain and product systems management for their applicability for catchment-based studies. The metrics are embedded in the systems model and are used for the quantification of the environmental outputs of alternative strategies at a catchment scale.

The work is driven by the UK national policy demands for integrated and resilient asset management. It is, however, relevant in an international context, as it contributes a novel approach for integrated water and asset management. It builds on a unique combination of concepts and methods that have never been applied before to serve asset management purposes. The research responds to the demand for approaches that allow for transparent reporting on the dependencies of the water sector on natural assets. The detailed mapping of catchment systems highlights areas of improvement for individual subsystems and enables the analysis of trade-offs, supporting decision-making.

2. The Fugue: a metaphor for the doctorate

The author's view of the doctoral research is that is can –metaphorically- be described as a fugue. Fugue (or Fuga) is a contrapuntal, polyphonic, sophisticated style of music composition in two or more voices (The Concise Oxford Dictionary of Music, Kennedy and Bourne-Kennedy 2013). It is built on one (or more) subject(s) - 'theme(s)', which are introduced in the beginning and recur frequently in the course of the composition, sounded successively in each voice. Although a fugue usually has three sections (exposition, development and recapitulation), many of their entities (e.g. episodes, tonic) are altered to serve the artistic outcome. In this sense, a fugue is a style of composition rather than a fixed structure.

The two compartments –fugue and the doctoral research- share structural characteristics. The theme of this research fugue is 'water'. The subject is sounded by four voices (aka disciplines): asset management, environmental science, sustainability and water policy. Each of them has a different and unique insight into the topic of water. The balanced combination of their 'views' and the careful selection of tools which represent their particular joints, will result to a holistic approach to water management at a catchment scale.

In order to achieve this balanced combination of the distinct voices –or viewpoints- without confusion, a structured approach to the research problem should be followed. A way to facilitate this process is to duplicate the unofficial framework (sequence of steps) as undertaken by a pianist who studies a music piece – in this case, a fugue. Practice has shown that the more structured the study approach, the more efficient it proves for the musician in terms of knowledge assembly throughout the process and time required to deliver the music outcome. This process resembles to constructing a musical jigsaw puzzle, piece by piece. Thus, when studying a fugue written for piano (aka two hands, 3, 4 or 5 voices) the steps undertaken are described as follows:

1. Assort the structural features (the metabolism) of the fugue. The main step is to mark the theme appearances, as the theme needs to be distinguishable and sounded throughout the fugue.
2. Assort the parts/notes sounded by each of the voices and then study each voice separately. This step finishes when one can perform each of the voices individually.
3. Combine the voices in pairs. For a fugue where four voices are sounded, the potential pairs/combinations are: Soprano and Alto, Soprano and Tenor, Soprano and Bass, Alto and Tenor, Alto and Bass, Tenor and Bass.
4. Study each hand separately. In this step, the accumulated work of the three previous steps proves very useful as it facilitates the process. It is like an 'evaluation' step, where one establishes the work of step 3 and adds to it several bits, if necessary. By the end of this step, one should be able to perform each single hand individually.
5. Combine the two hands to sound the fugue as a whole. In this step, all voices of the fugue are sounded together to deliver the musical outcome. It is essential to maintain the particular features of each of the voices comprising the music piece and to balance their sound. Further to this, special attention should be paid to the rendition of the structural features of the fugue: the theme should be sounded at each of its appearances, whilst the rest of the structural features (e.g. 'bridge') should be performed accordingly.

The use of the fugue metaphor to describe the undertaken research suggests the complexity of water-related research and that for the completion of the work a knowledge assembly from various fields –voices of the fugue- is required. Further, the meticulous and structured approach towards performing a fugue has been inspirational for the development of the methodology introduced through the research. It also inspired and formulated the individual steps undertaken for the creation of the linkages between or among disciplines, during the creation of the methodologies underpinning the research.

The following chapters aim to provide an insight at the diverse viewpoints relating to the theme of the fugue (water) and introduce the structure and building process of the comprehensive methodology (Chapter 5; Chapter 6). As such, a multi-discipline literature review follows (Chapter 2) and the research approach is presented (Chapter 3). The modelling schema created is tested on an example case study (Chapter 4) and its outputs (Chapter 6) are discussed (Chapter 7) to highlight the novelty of the work (Chapter 8) and reveal future research challenges.

Chapter 2: Literature Review

Chapter 2 sets the scene for the research undertaken. It provides an overview of the up to date literature on the topics relevant to the doctoral research. Also, a gap analysis is performed, aimed at highlighting the areas of improvement of the existing literature. The structure of the chapter reflects the rationale adopted and the steps undertaken throughout the research process.

The research outputs are policy-driven and are intended to inform the asset management strategies of the water sector. Thus, a policy requirement analysis is firstly performed, outlining the regulatory demands for the water sector in England and Wales and setting the foundation of the research. Then, the notion of asset management in introduced, in order to establish the terminological grounds of the work. The policy demands the development of systemic asset management strategies to drive the selection of life cycle management tools as the basis of the methodology created for the research. The knowledge gaps in regards to the scale and focus of both asset management strategies and life cycle management tools lead to the introduction of a new scale of focus for their joint application. The catchment (or watershed) is selected for the creation of a novel approach to strategic asset management in the water sector, as a scale where a holistic perspective on decision-making can be adopted.

2.1. Regulatory bodies & Policy Requirement Analysis for the Water Sector in England and Wales

The Water Sector in England and Wales (henceforth referred to as Water Sector) is among the most heavily regulated industries in the United Kingdom and is regarded by most observers as efficient and well managed (OFWAT 2006). Since the privatisation of the sector in 1989 (Water Act 1989), water companies in England and Wales have been privately owned. Although the type of ownership varies, the sector largely operates as a monopoly industry, with each water company covering a fixed geographical area. To ensure the interests of customers and the environment were secured, privatisation led to the reconstruction and, effectively, to the separation, of the roles of the regulation and provision of water and sewerage services (Hainsworth and Salvi 2014). Three separate, independent bodies were established to regulate the activities of the water and sewerage companies. These cover three main pillars: environmental regulation, economic regulation, customer provision. The Department for Environment, Food and Rural Affairs (DEFRA) monitors a wide spectrum of water issues via two mechanisms with distinct focal areas: the Drinking Water Inspectorate (DWI) which scrutinises the quality of drinking water and the

Environment Agency (EA) which controls issues related to environmental regulations and quality standards. Ofwat is the economic regulator of the water sector, whose role is to perform the balancing act between investment needs and affordable water bills and ensure the quality of the provided water services. The Customer Council for Water (CCWater) is not an official regulator, but rather a, independent public body representing the customers' views and offering impartial advice on water issues.

One of the main elements of water regulation is the price setting process, which is coordinated by Ofwat in five-yearly cycles, which are refereed as Asset Management Planning (AMP) periods or Price Review (PR) process. The annotation widely used to describe the price review process is PR followed by the year in which the prices are agreed. Currently, the water industry operates on the AMP6 (the sixth asset planning period since the privatisation of the industry) and on the plans, costs and charges agreed for the PR14. The PR14 initiated a more customer-oriented approach from water companies which encouraged service providers to focus on further developing what was considered as priority by their customers. It also introduced the 'Total Expenditure' (or totex) approach to cost and investment, moving away from the favourable financial treatment for capital expenditure (or capex). The totex approach enables the water sector to re-design their AMPs and shift their focus on both financial and natural systems in order to satisfy the priorities of different customer groups as identified in the extensive customer engagement programme launched by the sector as part of the 'outcome' concept strategy.

Over the years, a number of Parliamentary Water Acts outlined the duties and obligations of the 'water undertakers'. The strategies developed and adopted by the water sector were aimed at reflecting their statutory duties: predict customer needs, promote effective competition and ensure efficient finance for companies (OFWAT 2015a). Recently, the Water Act 2014 introduced the 'Resilience Objective' which is defined as "securing the resilience of both water systems and services, in the long term, without compromising the natural environment" and includes issues of both supply and demand. Interestingly, reliability of services, resilience and protection of ecosystems had also been identified as customers' priorities for the PR14 process (**Table 2.1.**).

Further consultation documents (OFWAT 2015b) reinforce the regulatory view on customer engagement and the maintenance of resilience for both financial and natural systems. The 'Trust in Water' or 'Water2020 Regulatory Framework' encourages water companies to design optimal options for resilience through strategic, regional, cross-sectorial planning. Water resources planning is placed at the centre of interest to achieve these optimal solutions, especially in regards to the market reform following in 2017. The development of information data bases for water accounts would provide new and better information on the water use and demand and serve as the basis for new conceptual frameworks for resource bidding options.

Table 2.1.: Summary of Policy Requirement Analysis. Resilience and the development of systemic approaches are identified as priorities for the regulatory bodies of the water sector in England and Wales. The consideration of ecosystems in strategic planning would enable the sector to meet their resilience duty, whilst responding to the European water regulatory demands.

Regulation/ Consultation Demand for	Water Framework Directive (EC/2000/60)	Price Review 14 (AMP6)	Water Act 2014	Consultation Resilience (Ofwat 2015a)	Trust in Water (Ofwat 2015b)	Roadmap to Resilience (Defra 2016)
Resilience		●	●	●	●	●
Systemic Approaches			●	●	●	●
Catchments	● (Phase 1)			●		●
Ecosystems	● (Phase 1)	●	●			●
Water Accounting	● (Phase 2)				●	●

The 'Roadmap to Resilience' report (Defra 2016) clearly states that the sector needs to undertake water resources planning frameworks adopting a long term, national view and embracing systems' resilience. The water sector is now incentivised to enhance the resilience of catchment systems, in an attempt to increase the water availability without putting the natural environment at risk. As part of the process of creating catchment-based frameworks, water companies need to develop structured and accurate databases for the catchments' water regimes. This information is essential to tailor approaches in order to manage access based on the consideration of whole-life costs and benefits. Further, the development of water accounts would enable the implementation of the second phase of the Water Framework Directive (WFD, EC/2000/60) and of the River Basin Management Plans (RBMPs). This includes the creation of an inventory of water resources and demands and of the water exploitation systems and their water balances in a consistent and structured format.

The policy requirement analysis (**Table 2.1.**) identifies the creation of systemic approaches suitable for strategic planning as a priority for the regulatory bodies of the water sector of England and Wales. The consideration of ecosystems through consistent water accounting and catchment-based frameworks is highly recommended as a means to meet their resilience duty. Due to the established process of Price Review, the Asset Management Planning process would enable the water sector to take the changes forward. The transformation of asset management is considered as the vehicle towards resilience and regulatory compliance.

2.2. Asset Management in the Water Sector

The Institute of Asset Management (IAM) defines an 'asset' as an item, thing or entity that has potential or actual value to an organisation. Although this definition is rather generic and wide, it mainly relates to physical assets (i.e. infrastructure). The Publicly Available Specification (PAS 55, BS ISO 55000:2014) -published by the IAM- defines 'Asset Management' as the "Systematic and coordinated activities and practices through which an organisation optimally and sustainably manages its asset systems, their associated performance, risks and expenditures over their life cycles for the purpose of achieving organisational strategic plan" (**Figure 2.1**.). PAS 55 has been used as a roadmap from the water industry –including Wessex Water Services Ltd- to develop their maturity and to maximise the ability to deliver their strategic objectives.

A variety of definitions exist for asset management (InfraGuide-FCM 2005, BSI PAS 55-1:2008, IIMM, 2011). Although varying in extent and scope, all serve to illustrate the different ways in which asset management can be understood and implemented by different organisations (Echelai 2013). Based on the guidelines provided, asset management has been developing over the past decades and is being used to form a structural framework to meet regulations and improve business efficiency. Each framework developed represents an organisation's understanding of the world and provides a frame for reference (Illaszewicz and Bradshaw 2013).

There is a consensus in all frameworks and general guidelines that Asset Management aims to provide a customer focus for business to systemically invest, maintain, upgrade and operate infrastructure assets. The primary goal of asset management is to meet a required level of service, in the most cost effective manner, through the management of assets for present and future. It involves the balancing of costs, opportunities and risks against the desired performance of assets and the organisational objectives.

Asset Management (AM) has become more important than ever before because it has emerged as a tool to support important decisions (Too 2011). It is also described as a tool that enables an organisation to examine the need for, and performance of, assets and asset systems at different levels. Additionally, it enables the application of analytical approaches towards managing an asset over the different stages of its life cycle (which can start with the conception of the need for the asset, through to its disposal, and includes the managing of any potential post disposal liabilities).

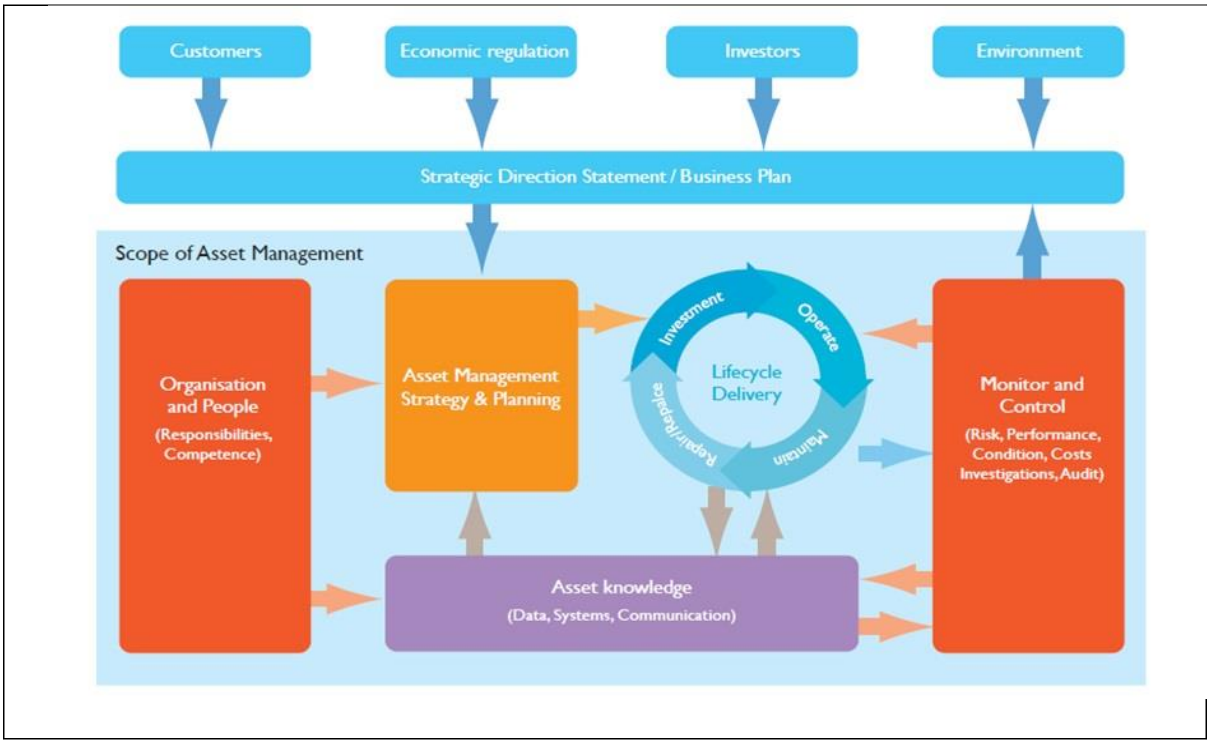


Figure 2.1: Conceptual Model of Asset Management according to the Institute of Asset Management. Abstracted from 'The Anatomy of Asset Management' (Public Available Specification-PAS 55).

The perception of an improved asset management involves enhanced service and customer satisfaction, improved governance and accountability, advanced risk management and financial efficiency and enables more sustainable solutions (Woodhouse 2006). Asset Management (AM) strategies have evolved from the specific conditions in which the organisation operates and can provide opportunities to formulate decisions which impact upon the success of the organisation's strategic goals (Kwok et al. 2010 a,b). In order to meet the special needs of the facilities in different industries, various approaches are available which enable the optimisation of the outcome (Kwok et al. 2010b). Palmer (2010) defines energy policy, climate change regulation, asset capital costs and strategic resources as the challenges faced by the UK water sector, which would affect future asset investments. Asset investment planning requires the provision of 'sustainable', novel solutions and a balance on the whole-life costs of assets, while complying with carbon and resource recovery regulations (Kwok et al. 2010a;b). To this end, case studies from water industries (Echelai 2013; van der Velder et al. 2013; Too 2011; van Heck 2010; Kwok et al. 2010a) have placed emphasis on refining asset management strategies implemented internally in each organisation to enhance their sustainability. In many cases, the novel, wider and integrated model introduced includes the risk of not meeting service requirements and standards in the maintenance and operation of built infrastructure.

Following the PAS-55 definitions for asset management and asset systems, water industries (e.g. Waternet and Rijkswaterstaat in Netherlands) have created frameworks based on systems thinking to provide better service to their customers. Provided that the service delivery of water companies heavily depends on the function and performance of physical assets, the boundaries of the aforementioned systems have been drawn around either their physical assets (i.e. capital, operation and refurbishment costs of infrastructure) or management and maintenance (i.e. reliable asset data, long-term performance based maintenance programmes).

As policy-related documentation reveals, for the water sector in England and Wales physical assets have acted as the centre of asset management activities. This approach has ensured their viability and the service delivery to their customers, while balancing whole life costs. Their planning has historically prioritised performance, maintenance and efficiency of their built assets systems (e.g. infrastructure). Further, under environmental and societal pressures (e.g. environmental regulations, statutory standards for service performance, stakeholders), their economic and planning strategies adopt abatement cost methodologies (e.g. paying fines) against the environmental burdens they provoke. Therefore, water companies have perceived their entities in isolation from the wider system to which they belong (i.e. their region of service). Prior to the implementation of PR14, investment decisions have been driven by their viability and ability to secure the provision of qualitative services to their customers. The efforts of the water sector to adopt sustainable principles has been restricted within the boundaries of their built environment, treating the wider environment as an externality to their asset systems.

More recently, the water sector of England and Wales has been officially encouraged (DEFRA 2016) to develop and broadly adopt novel systemic approaches that would enable joint planning for asset and water resources management. These approaches would enhance the resilience of both natural and physical (built) systems. Further to the policy consultation, a research report published by the UK Water Industry Research (UKWIR 2014) suggests that water companies should work collaboratively with stakeholders in order to fully consider and seek to achieve a balance between social and environmental costs (e.g. wider environmental impacts of carbon emissions, increased bills etc.). In other words, the sector is advised to adopt a more integrated approach in their investment plans and re-draw the boundaries of their asset systems. Defining their strategic goals around a different centre, for example the environment, or society, would ensure alignment with the national policy and international consultations (e.g. UN Natural Capital Declaration), whilst motivating stakeholders to share their principles and assist towards truly sustainable solutions.

The first milestone for re-designing asset systems for sustainable and resilient systems is the shift of scale. Research and policy shows evidence that the optimum scale for assessing water-related sustainability is that of the catchment (or watershed).

2.3. The *Catchment* as a System

This section defines the concept of the catchment and describes it as a system comprising of natural and artificial elements. It also introduces the term of Catchment as an Asset System and discusses its relevance to the creation of frameworks for the design of sustainable and resilient systems.

2.3.1. Defining a Catchment

There is a considerable heterogeneity and ambiguity in the literature with regard to the definition of the term 'catchment'. Recent studies (Godskesen et al. 2013; Yang et al. 2013; Angrill et al. 2012; Stoeglehner et al. 2011; Basset-Mens et al. 2006) carried out at this scale differ in terms of assumptions made about the spatial information contained within the delineated boundaries of a catchment. In these studies, the boundaries of a catchment are drawn around diverse entities and refer to groundwater aquifers, tributaries, hinterland areas or even rainwater harvesting systems.

In order to avoid terminological ambiguities, and guarantee the alignment with the current legislation (Water Environmental Regulations, No.3242, 2003) the term 'catchment' is defined from a hydrological point of view for the needs of the undertaken research.

Catchment is the area from which a surface watercourse or a groundwater system delivers its water. A surface catchment area may overlie an aquifer system, but may be unconnected with the aquifer rock itself if there are intervening impermeable aquicludes (Oxford Dictionary of Earth Sciences). The boundary between separate catchment areas or drainage basins is called 'divide', but it is also referred as 'watershed', in British language usage.

According to the definitions provided in the Water Framework Directive (WFD, EC 2000/60), the term 'river basin', stands for the area of land from which all surface run-off flows through a sequence of streams, rivers and possibly lakes into the sea at a single river mouth, estuary or delta. The fundamental unit (Article 3(1)) for its implementation, however, is that named 'river basin district', referring to the area of land and sea, made of one or more neighbouring river basins together with their associated groundwater and coastal waters.

Comparing and combining the aforementioned definitions, the 'river basin district' of the WFD is the hydrological 'catchment', since both refer to systems delivering both surface water and groundwater to a single river mouth, estuary or delta or to the sea.

2.3.2. Thinking the Catchment as a System

Catchment management is about using land in ways that benefit the water environment. Historically, in the UK, water authorities organised themselves at the river basin scale in order to control land use around water sources and prevent contamination of groundwater. However, after privatisation of the sector in 1989, the focus shifted to upgrading water and sewage treatment infrastructure to provide greater guarantees that drinking water and effluent standards would be met within short timescales (Rouse 2013).

Nevertheless, there has recently been an upsurge interest in catchment management, as a less resource-intensive way to protect water bodies (UKWIR 2014). An increasing number of water companies have focused on managing erosion and leaching affecting the quality of water in their service regions, with a particular concern regarding the nitrates and pesticides used on land and their impact on drinking water sources nearby. These strategies were driven by their commitment to the governmental River Basin Management Plans (RBMPs) which are designed to protect and improve the quality of the water quality, under the demands of the Water Framework Directive. It was the introduction of this legislative framework that established the catchment as the fundamental unit for managing water resources and highlighted the essence of integrated management of catchment systems for sustainable water management. A plethora of research studies dealing with the ecological quality of water courses or the development of integrated management plans have been published over the past decade. Further, the recommended use of qualitative impact assessment tools such as DPSIR (Driving force-Pressure-Solution-Impact-Response) model or SWOT (Strengths- Weaknesses-Opportunities- Threats) analysis, as part of the RBMPs, indicates that regulators acknowledge the complexity of the catchment as a system; thus, its societal extensions.

Indeed, the regional territory of a catchment consists of a number of natural, semi-natural and artificial landscapes, composed of a mosaic of interacting ecosystems (or subsets). Apart from the natural boundaries wherein the water-related ecosystem functions take place, catchments (or watersheds) have been characterised as pertinent spatial units for studying the interactions between humans and the environment (Billen et al. 2011) or the various types of capital (Pérez-Magueo et al. 2013) since drainage networks have historically acted as determinant factors of settlement location choice or agricultural and commercial activities. As such, the catchment can be described as a single integrated system which includes both natural elements (biosphere) and infrastructure (technoshpere). The sustainability pillars (environment, economy, society) co-exist and interact within its spatial boundaries (**Figure 2.2**). Creating approaches at a catchment scale would align with the vision of Sustainable Development (WCED 1987) which recognises that "social and economic progress should be simultaneous and integrated with the vitality of supporting ecosystems".

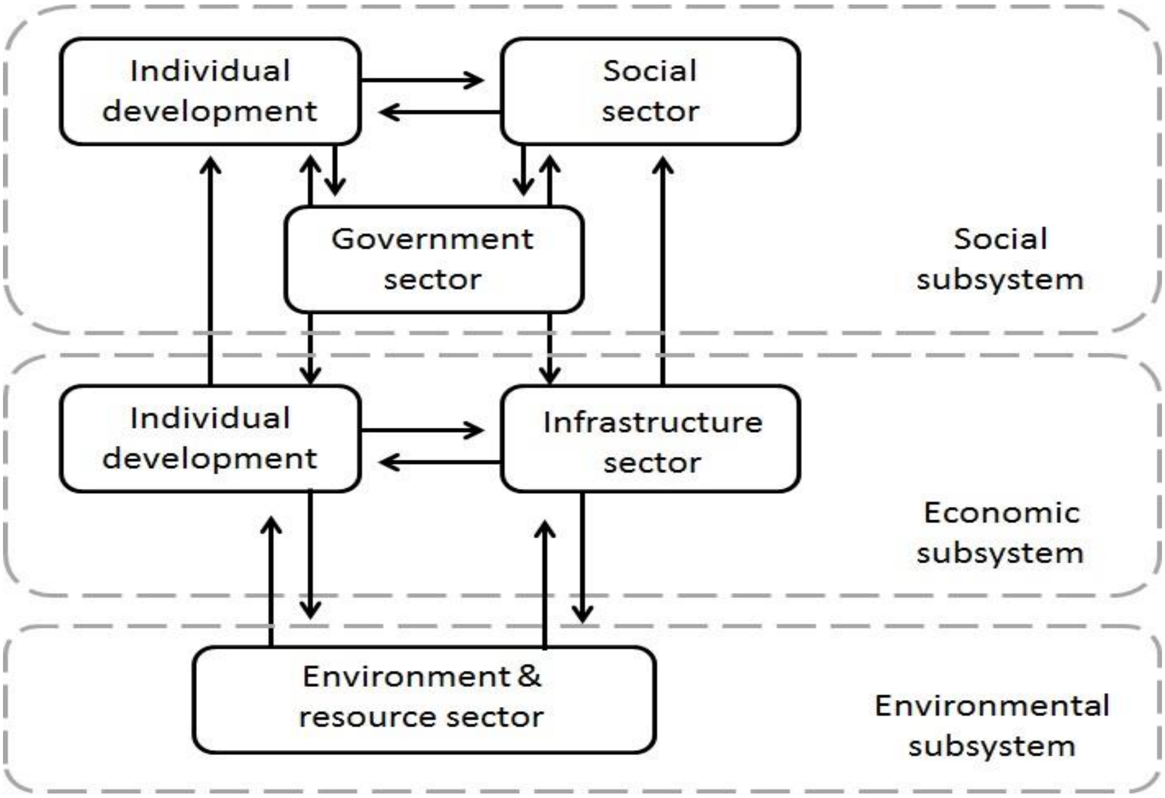


Figure 2.2.: The three sustainability pillars as subsystems of the catchment as a system. Abstracted from Hester and Little 2013.

There is a growing recognition that to meet the goal of sustainable and Integrated Water Resources Management (IWRM), there is a need for improved 'integrated' catchment management (ICM) (Macleod et al. 2007) and the design of local policies which involve alliances of a wide range of stakeholders (Prato and Herath. 2007). The concepts of IWRM and ICM are strongly interlinked, as the former has emerged in order to enable the achievement of sustainable management of water resources for a range of uses and several stakeholders (institutions, authorities, clients, population, and agriculture), while the latter provides a conceptual framework for solving water-related problems of multiple actors. ICM is not, however, a fixed-formula and requires different and creative conceptualisations of catchments and of their processes (Macleod et al. 2007; Toit 2005).

The essence of evaluating and assessing environmental sustainability of water at a catchment/watershed scale is argued in recent peer-reviewed literature (Nafi et al. 2014; Hester and Little 2013). In these works, it is argued that the catchment not only constitutes the fundamental unit for water resources analysis, but is also a scale ideally used by policy-makers.

The new paradigm of ICM implies the reorganisation of stakeholders that transcends sectorial boundaries (Nafi et al. 2014) and the use of appropriate scientific tools that would enable the integration between policies, science and their implementation. However, despite the impressive diversity of available measures for water management analysis, only a few recent studies address the importance of integrating the existing approaches into a unified framework for assessing sustainability at the watershed scale (Hester and Little 2013). The authors stress that sustainable approach to water requires integrated measures; that is, the combined use of non-integrated measures (such as components of the hydrological cycle or spatially explicit measures) within different scales or units. For a complete assessment of the environmental sustainability of water resources requires the use of 'common currencies' (referring to popular methods) is suggested, since the majority of the existing measures are suitable for independent quantifications and appear to be complementary.

The discussed literature shows evidence that in order to achieve the creation of an integrated catchment-based approach, the specific characteristics of the distinct subsets comprising the catchments along with their interactions among them should be identified and thoroughly studied. Sustainably managing an integrated system conveys the sustainable management of each of its elements, not only individually, but also as a whole, and thus, the creation of a 'systemic' approach. The interdependencies and interconnections among the elements of a system need to be identified in advance. The reason for studying each element separately and as part of the system lies on the rationale that any 'individual action' affects and reacts with the system (i.e. the 'whole'). Meeting an overall aim for a system (e.g. a catchment) means meeting specific objectives (i.e. optimising the function) for its individual components (e.g. infrastructure, land). The necessity for the integration of water utility systems in the frame of a catchment has been discussed (Everard 2012) as an essential for their capacity to support human well-being. Water infrastructure integrates multiple pressures from the catchment within it is built. It, therefore, becomes disproportionately vulnerable to climatic, hydrological, chemical, ecological and morphological pressures that affect its performance and service delivery.

In the River Basin Districts (RBD) of the UK, water quality issues relating to the diffuse pollution deriving from the agricultural sector have been identified as the main water environment challenge (Martin-Ortega et al. 2012). Significant improvements are needed to farm practices, in order to protect water quality. As a result, the water sector is encouraged to work collaboratively with stakeholders of the watershed areas under their service (UKWIR 2014), in order seek solutions for achieving balance between social and environmental costs (e.g. wider environmental impacts of carbon emissions, increased bills etc.) at a local scale. The creation of integrated, systemic catchment-based approaches will not only allow for the internalisation of elements which have been treated as externalities to date, such as land, but will also enable the assessment of their influence to the service of built assets. They would also support the consideration of stakeholders' requirements in the asset management planning process. The design of regional strategies based on integrated water resources and asset planning would enable the development of optimal solutions for catchment systems and the formulation of a more resilient water sector.

The principles of sustainable development underpin integrated catchment management. The creation of catchment-based approaches would require defining 'sustainability' for the given system, through the identification of the local stakeholders' needs and the definition of the interactions and models of cooperation among them.

2.4. Sustainability and Water Systems

The field of sustainability science is largely broad and diverse as evidenced by the number of review articles published over the years (e.g. Little et al. 2016; Miller et al. 2014; NRC 2014; Zaccai et al. 2012; Kuhlman et al. 2010). Undertaking another comprehensive review of the concept is considered out of the scope of research. Nevertheless, an overview of recent developments in the field would inform the content and shape the basis for further discussion. The focus of the quoted literature and the following discussion is geared on water resources and catchment systems, driven by the emphasis of the research on these topics.

2.4.1. Natural Resources & Environmental Impact Assessment

Resource efficiency is considered a key-element for sustainable development, as identified by the current political interest in the future availability of natural resources. Despite the initial, oft-quoted, 'Brundtland' anthropocentric definition of 'sustainable development' (WCED 1987) as 'meeting the needs of the present without compromising the ability of future generations to meet their own needs', the 'eco-centric' approach (O' Riordan and Voisey 1997) is prevailing (Jones et al. 2007), according to which the integrity of the biosphere underpins social and economic development.

Academic literature generally distinguishes natural resources between biotic and abiotic (Finnveden et al. 2009). According to Lindeijer et al. (2002), abiotic resources are inorganic or non-living materials at the moment of extraction (e.g. water and metals), while biotic resources are living at least until the moment of extraction from the natural environment (e.g. wood and fish). The latter does not include biotic resources reproduced by an industrial production process (e.g. livestock or agricultural crops). Another categorisation includes stock, fund and flow resources

(Klingamir et al. 2014). A schematic representation of natural resources and their categorisations is given in **Figure 2.3.** Resources could be evaluated in relation to their depletion (consumption related to geological/natural reserve), scarcity (economic availability of the resource) and criticality (a resource that is scarce and also crucial for society).

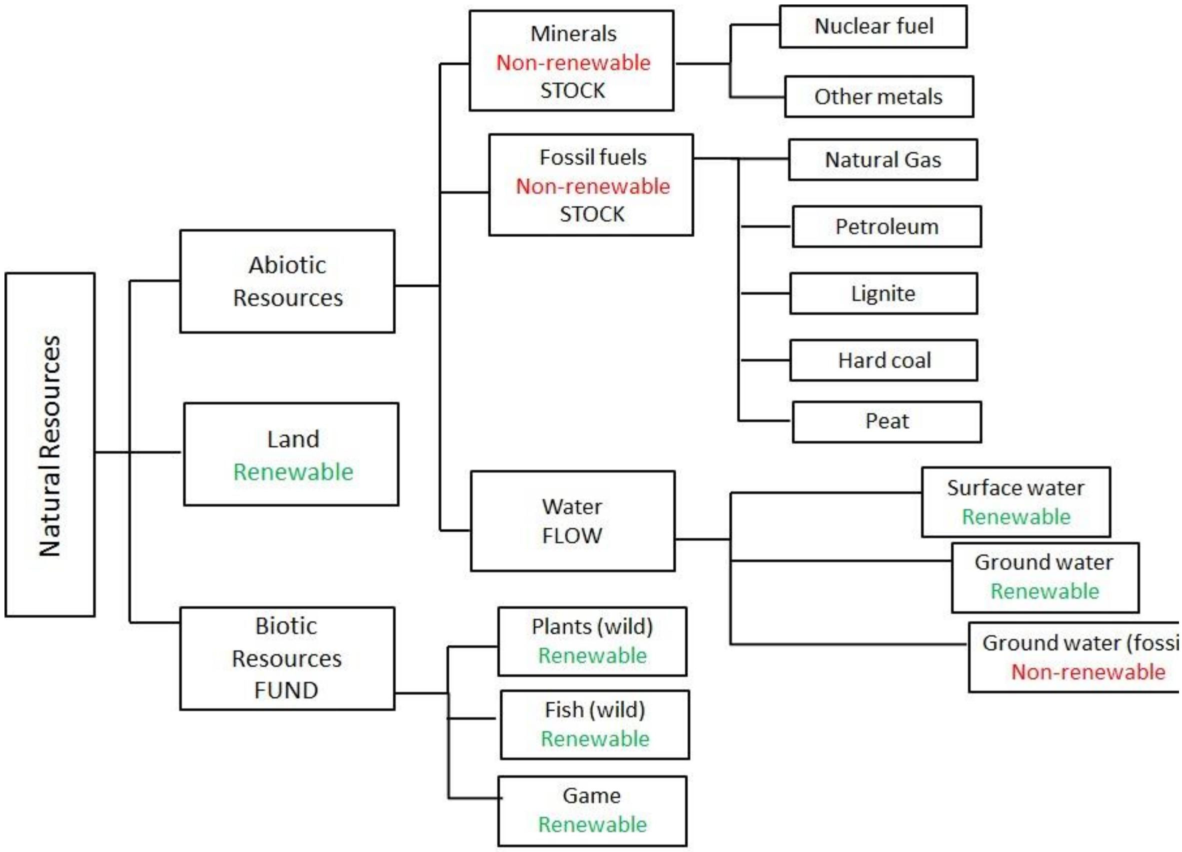


Figure 2.3.: A schematic representation of the classification of natural resources. Adapted from Klinglamir et al. 2014. Stock resources exist as finite amount in the natural environment (e.g. rock) or renewable rates on timescales too large to be compared with human rate of consumption (e.g. oil). Fund resources can be depleted at a rate dependant on a ratio of extraction to regrowth or to renewal rate (e.g. plants). Flow resources are those which cannot be depleted, but face the risk for temporal or spatial non-availability.

The operational rules to sustainability, as outlined by Daly (1990), underpinned and provided broad guidelines for the plethora of approaches developed to assess environmental sustainability (Little et al. 2016). Environmental Assessment (EA) refers to both environmental impact assessment (EIA) and strategic environmental assessment (SEA) (Jones et al. 2007; Wood 2003). EIA is a systematic and integrative process for considering possible impacts from a project significantly affecting the natural and man-made environment and takes place prior to the approval of a proposal. SEA is an equivalent process undertaken at the policy, plan or programme level.

Past works (Morgan 2012; Jones et al. 2007) suggest that although the proliferation of diverse forms of impact assessment reflects the recognised value of a structured and consistent approach to evaluating environmental aspects in decision-making, there are several outstanding challenges for EA. These relate to the necessity of the forms of impact assessment to contribute to the scope of sustainability and as such, to be grounded on well-defined principles and be conducted in an integrated way. A recent review article (Little et al. 2016) evaluates the range of sustainability assessment approaches and compares them to the nature of the sustainability problem, aiming to establish whether the available approaches are appropriate for the task for which they have been developed. The authors categorise the existing suite of EA approaches into two main categories: (a) design-based, which generally follow principles or guidelines and (b) approaches that employ computational frameworks and/or indicators. The former category includes frameworks which examine the factors causing impacts, whereas the latter are used to assess the effects of the impacts.

From the suite of frameworks and tools presented and analysed in the recent review articles, those of particular interest for the undertaken research are those based on whole-system design and integrated approaches.

2.4.2. Integrated Sustainability Assessment

The complexity of the sustainability exercise has driven the formulation of approaches that enable the integrated assessment for evaluating environmental science, technology and policy problems (Laniak et al. 2013). Integrated Approaches employ scenario analysis to characterise hypothetical future pathways. The nature of the scenarios (qualitative or quantitative) classifies the approach under the categories analysed above (section **2.4.1.**). Whole-system design enables the integration of sustainability principles and thinking in engineering, especially in the definition of causalities and interdependencies (Blizzard et al. 2013; Charnley et al. 2011; Stasinopoulos et al. 2010).

Little et al. (2016) discusses that there is a critical need for coupling whole-system approaches with integrated environmental assessment, in order to understand the behaviour of complex systems and the relations across their environmental, economic and social compartments. A unified ‘systems-of-systems’ approach is suggested, as a means to enable the endeavour. To date, the fragmented field of sustainability hampers the coupling. Nevertheless, several attempts have been pursued toward this end, with special focus on the integration of the environmental and economic aspects of sustainability assessment for urban (e.g. Ma et al. 2015), energy (e.g. Rudell et al. 2014) catchment (e.g. Avila-Foucat et al. 2012) and aquifer (e.g. Kahil et al. 2016) systems.

Among the approaches developed to date, the ‘capital approach’ to sustainable development has gained popularity over the years, as a common framework of communication among countries and institutions and is considered promising for achieving consensus on the issue of sustainability and policy-making (Kulig et al. 2010). The capital approach is underpinned by the aforementioned 'Brundtland' definition (WCED 1987) of 'sustainability' and is firmly based in macro-economic theory (Kulig et al. 2010). For the purpose of sustainability assessment, four types of capital are distinguished: economic, natural, human and social (UNECE 2009). The approach enabled the expansion of the notion of 'capital' beyond economics and the measurement of capital stocks in non-monetary terms.

Based on the capital approach and the System of National Accounts (SNA), the United Nations (UN) introduced the System of Environmental and Economic Accounting (SEEA) (originally in 1993 and the revised version in 2003), in an effort to incorporate environmental information into the national accounts. In 2012, the SEEA Central Framework was adopted by the United National Statistical Commission (UNSC) as an international statistical standard. It is designed to complement and extend the accounting of SNA and has widely used for the compilation of a set of interrelated accounts to record economic activity (Hein et al. 2015). It allows the integration of physical data about the environment in the SNA through the development of accounts that describe the supply and use of materials and energy, as well as the residuals and return flows generated.

The adoption of the SEEA Framework has driven innovations in environmental accounting and generated the development of 'Ecosystems Accounting' as a comprehensive and consistent framework for recording changes in ecosystems and their implications to people (Obst et al. 2014). As analysed by Hein et al. (2015), Ecosystem Accounting differs from various other ecosystem valuation approaches and enhances the SEEA Framework as it offers an integrated approach to analysing ecosystem assets or natural capital (i.e. the set of renewable and non-renewable environmental assets that directly or indirectly produce value or benefits to people).

The terminology and rationale introduced through the environmental expansion of SNA underpins policy recommendations by a number of international agencies. The Natural Capital Initiative (World Forum for Natural Capital, 2015) relates the poor management of the natural environment with catastrophic consequences on ecosystems productivity, human wellbeing and financial resilience. It strongly supports the adoption of the United Nations Natural Capital Declaration (NCD, UNEP 2012) which requests from financial institutions of the private and commodity sectors to integrate Earth's natural assets in their reporting, accounting and decision-making.

A systemic approach for the development of water-specific accounts was introduced by the United Nations Statistics Division (UNSD) in 2012, allowing for the performance of the Water Accounting exercise.

2.4.3. Water Accounting

In the absence of a standardised definition, the notion of Water Accounting (WA) can be described (according to the Bureau of Meteorology of the Australian Government, BoM 2014) as the systematic process of identifying, quantifying, reporting and publishing information about water as a resource (namely its sources and uses). The outputs of this exercise need to be demonstrated in a coherent format in order to ensure their functionality and suitability for decision-making in the water sector. Water Accounting has emerged as an appropriate tool to improve transparency and control in water management and assist in achieving the goals of integrated water resources management (Pedro-Monzonis et al. 2016b; Momblanch et al 2014).

Several WA methodologies have been developed by states and international organisations, with diverse focal points and presentation formats (Momblanch et al 2014; Gan et al. 2012). WA methodologies focus on the relationship between water use and economy (Ward and Pulido-Velásquez 2009), on the development of physical water accounts aimed at conflict resolution (Allan 2012), on the assessment of water productivity at different spatial scales (Karimi et al. 2013; 2012) or on the water uses for resource allocation purposes (AWAS, BoM 2012).

Further, the United Nations SEEA Framework has been recently expanded to include accounting for water flows. The System of Environmental Economic Accounting for Water (SEEAW) (UNSD, 2012) which provides a method of organising and presenting information relating to the physical volumes of water in the environment, water supply and economy (Pedro-Monzonís et al. 2016a; Vardon et al. 2007) and is based on the principle of the conservation of mass (sums of inflows equals the sums of outflows) (Molden and Sakthivadivel 1999). Currently, SEEAW is the most widespread hybrid water accounting approach with application in many counties, expanding from China and South Africa to a number of European countries (Pedro-Monzonís et al. 2016b, Tilmant et al. 2015; Momblanch et al 2014). Although SEEAW is displayed as a tool to build water balances at a river basin scale, there are concerns about its practical use by policy-makers. These are related to the lack of common definitions and procedures to build the water accounts and to the vast amounts of data required to achieve this. Methodological weaknesses of the method are also highlighted: it does not allow for comparisons for different territories and periods and does not explicitly account for the environmental requirements of the catchment (Pedro-Monzonís et al. 2016a; Vicente et al. 2016; Tilmant et al. 2015; Dimova et al. 2014). The use of hydrological and hydraulic models is highly recommended (Pedro-Monzonis et al. 2016b; Vicente et al. 2016) as a tool to fill in the water accounts, especially those referring to complex natural hydrological processes (e.g. evapotranspiration, soil moisture, exchange between water bodies).

The growing volume of academic literature and the augmented research interest around Water Accounts has been mainly boosted because of policy necessity. As a requirement of the implementation of the Water Framework Directive (WFD, 2000/60/EC) the Member States, including UK, were required to design and put in action the River Basin Management Plans (RBMPs). These needed to include details on the site setting of the river basin (registry of the protected areas, environmental pressures affecting the water status, environmental targets, cost recovery) and a programme of measures to address issues. Additionally, RBMPs demanded for the inventory of water resources and demands, the regime of environmental flows of the catchment, its water exploitation systems and their water balances. The Water Blueprint (EC 2012), presented a strategic approach towards the implementation of the WFD and suggested the joint analysis of water policy objectives with the economic growth of multiple sectors in terms use as a way to improve the WFD water efficiency goals. The development of water accounts is currently one of the next steps to be implemented in the RBMPs and their development is considered as a tool to achieve the objective of water efficiency (Pedro-Monzonís et al. 2016a). Case studies and analyses (Vicente et al. 2016; Momblanch et al. 2014) suggest that the optimum scale for the implementation of existing water accounts is that of the river basin or catchment. This is particularly important for the studies focussing on the physical water accounts. The further division of river basins into smaller units has proven to introduce high level of uncertainty, especially for those systems with a low spatial variability (Vicente et al. 2016).

Further developments in the field of Water Accounting have emerged in the academic literature over the recent years in the form of Water Inventories. These approaches employ computational frameworks and indicators, which largely relate to the concept of ‘Water Footprint’ and the field of Life Cycle Management.

2.5. Life Cycle Thinking

In the context of progress of sustainability science, Life Cycle Thinking (LCT) may play a crucial role (Sala et al. 2013a,b). The prevalence of LCT in research, industry and policy has resulted in a vast volume of articles published during the last decade in this diverse field, as discussed by McManus and Taylor (2015). Applying LCT offers a way of incorporating sustainable development in decision-making processes (Valdivia et al. 2013). This means going beyond the traditional introverted focus of industries and taking into account the environmental, social, and economic impacts of a product/activity over its entire life cycle and value chain. In order to deal with the complexity involved in this endeavour, it is required to enhance the methodologies for integrated assessment and mainstreaming of LCT from product development to strategic policy support (Sala et al.

2013a). The most enhanced sustainability frameworks have recently been reviewed (Sala et al. 2013b) while a framework based on LCT has been proposed.

Life Cycle Sustainability Assessment (LCSA) refers to the evaluation of all environmental, social and economic negative impacts and benefits of a product throughout its life cycle and to the use of the result to support decision-making processes (UNEP/SETAC 2011). The idea of combining three LCT techniques into an LCSA framework was first formulated by Klöpffer (2008). The following equation expresses its concept and introduces its rationale: the assessment of the sustainability performance of a product should be carried out by the contemporary implementation of the three life cycle techniques (Valdivia et al. 2013).

LCSA = (environmental) LCA + LCC + S-LCA

(LCA=Life Cycle Assessment, LCC=Life Cycle Costing, S-LCA=Societal Life Cycle Assessment)

LCSA is a transdisciplinary integration of models, rather than a model itself (Guinée et al. 2011). It is a framework for looking from one viewpoint to specific sustainability questions, which demands the integration of disciplinary methods and tools for addressing the formulated questions. Structuring, selecting and linking the plethora of models practically available in relation to different types of life-cycle based questions is the main challenge of its application.

Literature (Sala et al. 2013a,b; Valdivia et al. 2013) suggests that the application of LCSA could benefit consumers, businesses and decision-makers in several ways. It would clarify the trade-offs between the three sustainability dimensions, life cycle stages and impacts and raise credibility by communicating useful quantitative and qualitative information regarding processes/products/strategies. Moreover, LCSA could support decision-makers in prioritising resources and investments and making sustainable choices, in terms of technologies and products. It could broaden the scope of Life Cycle Management to cover all three dimensions of sustainability (people, planet, prosperity) and to questions related to specific sector or even economy-wide levels or behavioural relations (Guinée et al. 2011). According to Sala et al. (2013a,b), LCSA should be developed in order to represent the holistic approach which integrates, rather than substitutes, the reductionist approach of the single part of the analysis. To achieve this, a balance between analytical and descriptive approaches towards a goal and solution-oriented decision support methodology should be maintained.

It is suggested (Swarr et al. 2011) that LCSA or the combination of the well-established LCT tools of Life Cycle Assessment (LCA) and Life Cycle Costing (LCC) is applied in case studies in order to gain experience and validate the utility of the methods across different sectors.

2.5.1. Life Cycle Assessment (LCA)

Life Cycle Assessment (LCA) is a technique used to quantify the environmental impacts associated with all the stages of a product, service or process from cradle-to-grave. It has gained popularity as a sustainability assessment method (Guinée et al. 2011), as evidenced by the increasing number of publications and databases supporting its implementation.

An LCA study must be carried out in accordance with the technical norms established by the ISO standard (ISO 14040 and ISO 14044). In the umbrella document, LCA is defined through the procedure for performing an LCA (**Figure 2.4.**). The standard for LCA also lists the following applications: identification of improvement possibilities, decision making, choice of environmental performance indicators and market claims (ISO 14040, 2006).

The strength of LCA is that it studies a whole system. The results are related to the function of the system, which allows comparisons between alternatives (Baumann and Tillman 2004). It is an engineering tool in the sense that technical systems and potential changes to them are studied. On the downside, the environmental impacts cannot be modelled at a very detailed level, since LCA is not site specific. In addition, economic and social aspects are normally not included in LCA; and neither is risk.

As illustrated in **Figure 2.4.**, the LCA procedure includes four main phases: goal and scope definition, inventory analysis, impact assessment interpretation (ISO 14040 2006). In brief, in the goal and scope definition phase, the product to be studied and the purpose of the study are determined. The scope affects the definition of the system boundaries and the level of detail. Life Cycle Inventory (LCI) analysis involves the collection of the data required and the framing of a flow model of a technical system according to the requirements of the goal and scope definition. Life Cycle Impact Assessment (LCIA) indicates/describes the impacts of the environmental loads quantified in the inventory analysis and establishes their relations. Finally, the interpretation phase summarises and discusses the results. While LCA is an iterative assessment, interpretation is required throughout all its phases to ensure rigorous outcomes. Further details regarding the LCA model and procedure can be found on the relevant standard document (ISO 14040 2006).

The value and usefulness of LCA heavily depend on the choices of methodologies made throughout the process (Settanni et al. 2012a). The four more critical choices of methodology for a researcher performing an LCA study are the definition of functional unit, system boundaries and allocation procedure, type of data used and impact assessment (Baumann and Tillman 2004).

The functional unit corresponds to a reference, quantitative flow to which all other modelled flows of the system under study are related. The principles of system boundary definition and allocation are decided during the goal and scope definition. Nevertheless, the methodological choices in LCA depend heavily on the questions or hypotheses formulated. The goal and scope definition phase is important since the appropriate LCA method depends on the purpose of each study (Finnveden et al. 2009).

Finnveden et al. (2009), Curran et al. (2005), and Baumann and Tillman (2004), have concluded in a distinction between two types of methods for LCA: attributional and consequential (**Table 2.2**.). Attributional LCA (aLCA) is defined by its focus on describing the environmentally relevant flows to and from a life cycle and its subsystems and its character is retrospective. Consequential LCA (cLCA) is defined by its aim to describe how environmentally relevant flows will change in response to possible decisions and is, by nature, more prospective (Finnveden et al. 2009; Curran et al. 2005). Other terms have been used to denote the two types of LCA, such as descriptive/accounting and change-oriented respectively. The relevance of the two LCA types for decision making, learning processes and modelling of future systems is also argued (McManus and Taylor 2015; Finnveden et al. 2009; Curran et al. 2005). Despite the debate on the suitability and applicability of the two types of LCAs for different research purposes, recent literature (Rajagopal 2016; Yang 2016) stresses the complementarities of the two LCAs and highlights the value of aLCA as a structural basis for the further development of the cLCA as a technique to explore the wider changes of an overall system (McManus and Taylor 2015) or as an approach for converging LCA with economic models (Earles and Halog 2011).

The distinction between the types of LCA indicates how the goal and scope definition stage influences critical methodological and data choices. In the same vein, Guinée et al (2002) make a similar distinction for LCA types based on three main types of questions: (a) accessional choices, (b) structural choices and (c) strategic choices. The different types of decision may require different modelling and data types or different scales in terms of time and impacts.

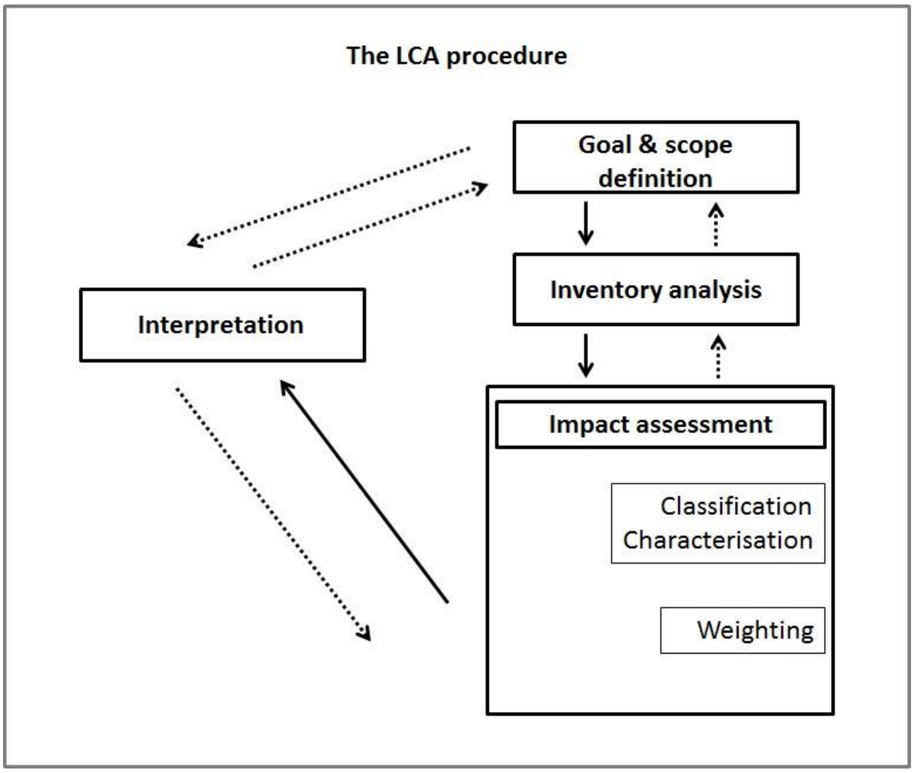


Figure 2.4.: The LCA procedure. The boxes indicate procedural steps and the arrows the order in which these are performed. Broken arrows indicate possible iterations. Adapted from Baumann and Tillman (2004).

Table2.2..: Characteristics of accounting and change-oriented LCAs Adapted from Baumann and Tillman (2004) and modified according to more updated literature (Finnveden et al. 2009).

	Type of LCA	
Characteristic	**Attributional**	**Consequential**
System boundaries	Additivity	Parts of system affected
	Completeness	
Allocation procedure	Reflecting causes of system	Reflecting effects of change
	Partitioning	System enlargement
Choice of data	Average	Marginal
System subdivision	–	Foreground & background

Traditionally LCA studies have treated water as an input flow, without differentiating water types or water quality criteria. More recently, the World Business Council on Sustainable Development (WBCSD) UNEP/SETAC Life Cycle Initiative was launched, focusing on the development and standardisation of tools and methods to assess freshwater use at different scales within the LCA framework. As a result, the water footprint standard (ISO 14046:2014) has been released, providing principles, requirements and guidelines for conducting and reporting water footprint assessments within the LCA framework. The development of methods to address water assessment is making considerable progress today, as witnessed by the increasing number of relevant published papers and reports.

A more detailed discussion on the relationship between LCA and water systems will follow (chapter section **2.6.**).

2.5.2. Life Cycle Costing

The economic counterpart of LCA is Life Cycle Costing (LCC). According to the relevant BS ISO standard (BS ISO 15656-5:2008), Life Cycle Costing (LCC) is a methodology for a systematic economic evaluation of life-cycle costs over a period of analysis, which could cover the entire life cycle or selected stages. In other words, LCC is a cost assessment of a product's cradle-to-cradle costs and is a way of accounting the total costs of built assets (e.g. equipment, infrastructure), aiming at estimating the cost associated with the existence of a product for comparing alternative products (Rebitzer and Hunkeler, 2003). Governments, organisations and industries have developed LCC methodologies in order to understand cost-drivers of a product system, to identify improvement options and to validate pricing strategies (Swarr et al. 2011).

Based on number of case studies, varying in goal and scope settings, as well as in methods and methodological choices, the UNEP/SETAC-Europe working group has identified three types of LCC: conventional LCC, environmental LCC and societal LCC (Ciroth et al. 2008). The system boundaries and the costs included in each type are presented in **Figure 2.5.** .

Briefly, Conventional LCC assesses all costs associated with the life cycle of a product that are directly covered by a single actor, focusing on real, internal costs. Environmental LCC assesses all costs associated with the life cycle of a product covered by one or multiple actors, including externalities that are anticipated to be internalised in the decision-relevant future. It provides an economic counterpart to the environmental metrics obtained from an LCA (Settanni et al. 2012b) and enhances conventional LCC by requiring the inclusion of all life stages and separate non-monetised LCA results. Societal LCC includes all of environmental LCC plus additional assessment of further external costs, usually in monetary terms.

Although most applications of life cycle frameworks include elements of environmental cost, there is scope for confusion, while most view life cycle costing as referring only to private (internal) costs (Ciroth et al. 2008). Those are the costs carried by a directly involved stakeholder and included in the price paid by the end user. Nonetheless, in more holistic concepts (e.g. environmental/societal LCC), external costs (externalities) are also included in the assessment. Externalities refer to cost induced to stakeholders outside the economic considerations of the system boundaries, which are internalised as real money flows and not included in the price paid by the end user.

Current challenges in systems thinking and sustainability science have pushed towards concepts suitable for an assessment of the economic implications of a product life cycle in a consistent sustainability framework; thus, would ensure LCC in an approach to estimate the economic dimension of sustainability. Works on the harmonisation of the set-up and principles of LCA and LCC (Heijungs et al. 2013; Swarr et al. 2011; Ciroth et al. 2008) show evidence of the existing methodological challenges in LCA and LCC integration. Nonetheless, in practice, the approach that has prevailed so far is the combination of LCC and LCA as separate yet consistent tools, thus excluding the integration of the former into the latter (Settanni et al. 2012a,b).

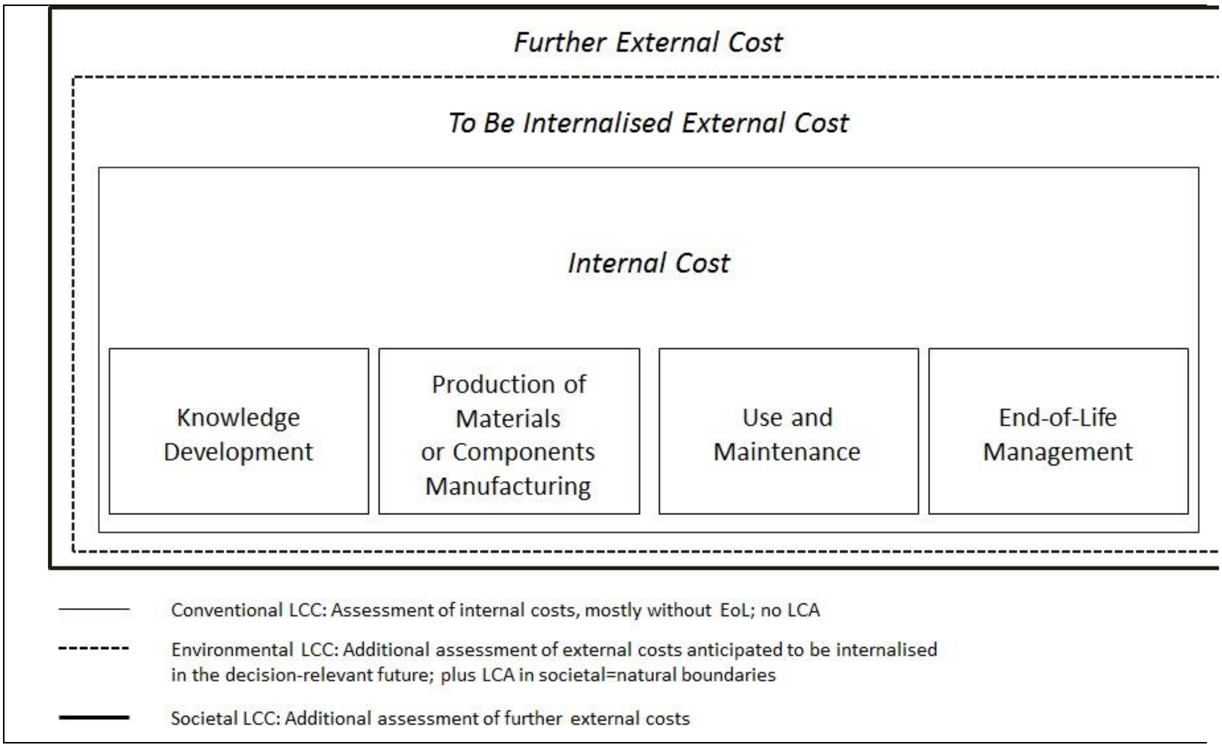


Figure 2.5.: System boundaries and costs included in the three types of Life Cycle Costing. Abstracted from Ciroth et al. (2008).

Detailed technical guidelines for conducting -environmental- LCC studies, as well as its joint use with LCA could be retrieved in recent UNEP/SETAC books (Swarr et al. 2011; Ciroth et al. 2008). These guidelines allow flexibility to adapt according to the specific needs of the each single case. A guiding principle is that the rigor of the economic analysis should be consistent with the goal and scope defined in the environmental analysis of the study.

The existing application of LCC in water systems research will be analysed in a following chapter section (section **2.6.**).

2.6. Life Cycle Management and Water Systems

This section discusses the relevance of the Life Cycle Management tools to the water sector and the wider water systems. Due to the wide application of life cycle thinking, the articles included in this section have been selected in order to indicate research trends and gaps.

2.6.1. Life Cycle Management in the Water Sector

Life Cycle Assessment (LCA) has proved well-suited for application in the water sector and has been characterised as a particularly useful tool for organisations wishing to look holistically to the environmental impacts, investigate alternative solutions and go beyond regulatory compliance (Barrios et al. 2008; Narangala and Trotter 2006). In the water industry LCA has been applied at a strategic and/or regional level, at project and process level and at a very specific level (Friedrich et al. 2007).

The application of LCA for industrial case studies has gained interest over the last few years. Recent industrial case studies (Bernard et al. 2014; Risch et al. 2014; Slagstad et al. 2014; Barjoveanu et al. 2014; Niero et al. 2014; Venkatesh and Brattedø 2010) adopt life cycle methods to assess environmental impacts of the urban water cycle. In a large and growing body of literature, the system boundaries of the undertaken LCA have been expanded in order to include the whole urban system (Yoshida et al. 2014; Chang et al. 2014; Slagstad et al. 2014; Barjoveanu et al. 2014; Lemos et al. 2013), i.e. freshwater abstraction, water treatment and production of tap water, water distribution, wastewater transport to the plant. Thus, the Urban Water Management System (UWMS) is selected as focal system for the undertaken assessments. A UWMS is a multifunctioning combination of decentralised sub-systems, representing the urban part of the water cycle (**Figure 2.6.**) (Nafi et al. 2014) and as such, it is perceived and managed as an integrated system, rather than separate units of infrastructure. In a more traditional approach, several studies assess different parts of the urban water cycle, focussing on the water (e.g. Bonton et al. 2012) or wastewater (e.g. Zang et al. 2015) treatment processes alone. Few other studies expand their boundaries investigating opportunities from waste by-products (e.g. sludge) in agriculture or energy production (Eriksson et al. 2014; Niero et al. 2014; Sadhukhan 2014). These provide evidence that research is moving from the plant to the river basin scale and a broader perspective is adopted in few works; that of the integrated management (e.g. Mouri et al. 2013a,b).

In the more technical aspects, the majority of the industrial case studies presented above implement consequential (change-oriented) LCAs. The urban water cycle is, therefore, divided into background and foreground systems. The functional unit selected does not alter through the stages and is adjusted to fit the scope of each case. Those targeting the water urban cycle as a whole (e.g. Yoshida et al. 2014; Bernard et al. 2014; Slagstad et al. 2014; Barjoveanu et al. 2014; Lemos et al. 2013) chose the production or consumption of one cubic meter of potable/tap water as their functional unit, whereas the studies focusing on treatment processes (e.g. Zang et al. 2015; Bonton et al. 2012) select chemical-related functional units, population or legislation dependant. In the majority of the case studies, two environmental mid-point impact assessment methods were selected: ReCiPe and CML, while the Ecological Scarcity 2006 end-point method was additionally used as well (Barjoveanu et al. 2014). Regarding datasets, the Ecoinvent database was selected in the majority of the case studies and local-specific or industrial data were used where applicable or necessary.

Regarding their content, case studies address mainly problems of local scale or concern, namely a city or settlement in different regions worldwide (Romania, Denmark, Singapore, Norway, Portugal) or a catchment area. LCA is used as a tool either to identify hotspots (Slagstad et al. 2014; Barjoveanu et al. 2014) in the water services system or to compare improvement alternatives in terms of selected treatment technologies (Risch et al. 2014; Niero et al. 2014; Mouri et al. 2013a,b) and implementation strategies (Bernard et al. 2014; Jeppsson et al. 2014). Water and wastewater treatment plants are investigated in terms of energy and chemical consumption, reaching in several cases the conclusion that in practice urban water utilities would have to perform a trade-off between the consumption of energy and chemicals and the discharge of pollutants to the environment (Slagstad et al. 2014). For the background system of the urban water cycle, tap water production (including water abstraction and treatment) is identified as the most energy demanding stage (Bernard et al. 2014; Barjoveanu et al. 2014; Lemos et al. 2013), whilst intensive energy consumption is embedded in the distribution of water (pipeline networks) as well (Slagstad et al. 2014; Lemos et al. 2013). The most important impact categories related to the urban water cycle -as identified in recent case studies- are the global warming (where WWTPs have the biggest contribution) and eutrophication of freshwater and marine water bodies. What is often commented and highlighted though is that environmental impacts of urban water systems are site-specific, as they depend on several local factors. Therefore, results obtained for a certain geographical area cannot be extrapolated to other areas (Risch et al. 2014; Lemos et al. 2013).

The research gaps which need to be addressed in the application of LCA on water-related systems and assessment have been identified and discussed in a recent review article (Corominas et al. 2013). The authors stress the need for the unanimous expansion of the goal and scope of LCA studies to include the entire urban water cycle, but also resources depletion and recycling options. Further, the LCA models need further advancements in order to adapt to the challenges occurring from the expansion of the scope: they need to include new target compounds, such as micropollutants in sludge, and advance or create the characterisation factors to address problems at a regional scale. The suggested advancements in combination with the need for reducing the uncertainty of the results, stress the need for improvements in data quality and data sharing options. This could be facilitated through the enhancement of stakeholders' participation and the strengthening of the LCA links with costing and societal aspects aiming to complete the whole picture of sustainability.

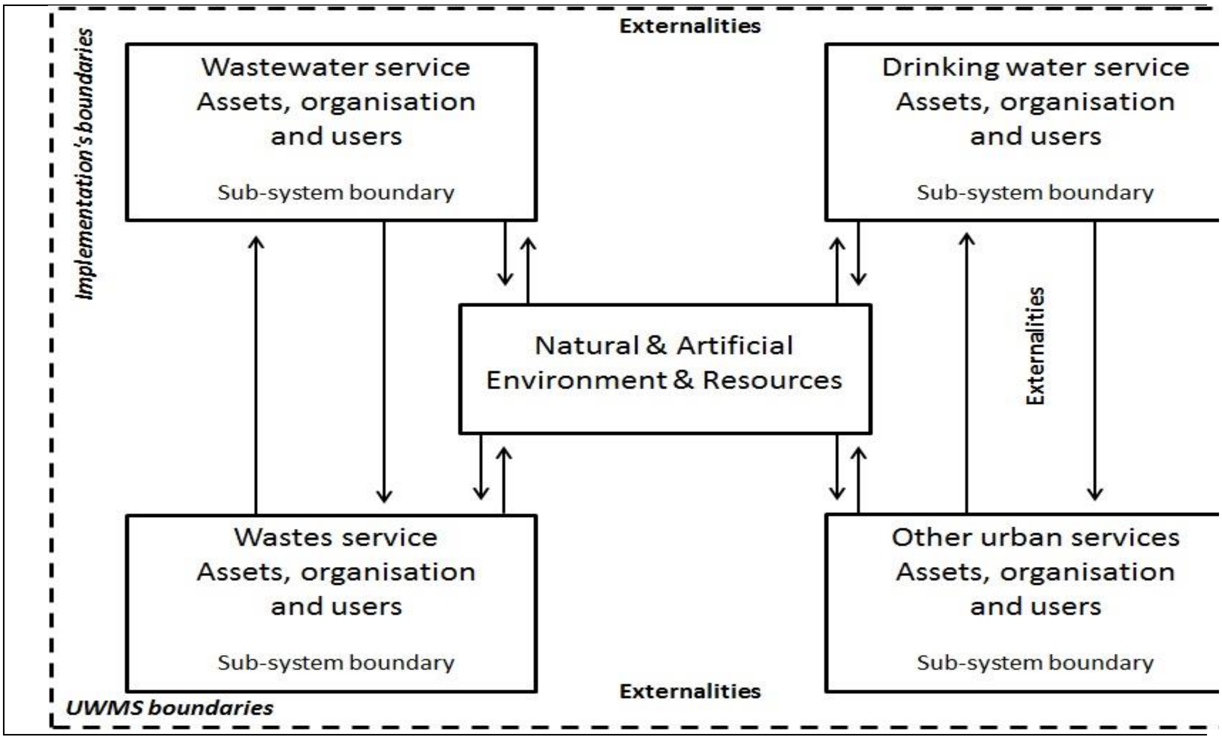


Figure 2.6.: The boundaries of an urban water management system (UWMS) and its implementation boundaries (e.g. watershed). Adapted from Nafi et al. (2014). Sub-systems fulfil several standard technical functions, while externalities include additional ones, e.g. environmental conservation or prevention of pollution.

To this end, the performance of life cycle management tools at an UWMS level could serve the goal of integration of those elements considered as externalities from a single-actor perspective (water industry), such as environmental conservation. An integrated life cycle management system is a prerequisite for demonstrating the benefits of strategies adopted, because, to date, the costs are isolated and addressed in fragmented ways across various actors (Nafi et al. 2014). Few recent case studies include a costing assessment as well. However, an explicit, joint LCA and LCC study has not yet been published. Therefore works to date perform costing analyses to identify the financially favourable option from a selected perspective (Bernard et al. 2014) or make use of economic valuation techniques (e.g. Willingness to Pay-WTP) either as weighing factors to the LCA results (Wang et al. 2013a) or as a value indicator (Mouri et al. 2013b). As case studies fail to combine environmental and economic assessment of services and policies related to the urban water cycle, other works attempt to develop costing tools and concepts applicable in synergy with the environmental life cycle assessment or shed light on methodological ambiguities. Thus, Jeppsson et al. (2014) develop a decision support tool to be used in the evaluation of control/operational strategies in water industry. They introduce a 3D graphical representation that shows interactions among effluent quality, operational cost and greenhouse gas (GHG) emissions, pointing out the importance of considering the existing interactions between the different stages of the urban water line. This 3D model graphic fits the concept of the portofolio presentation as suggested for the environmental and societal life cycle costing (LCC) results (Ciroth et al. 2008). Another study (Igos et al. 2013) develops a novel cost performance (CP) indicator, aiming to fairly compare water production plants. The rationale of monetised environmental assessment results lies on the fact that they can be easily communicated to decision makers and this work highlights the meaningfulness of using monetised LCA results in comparison with operational costs. Igos et al. 2013 make use of two monetisation methods: Eco-costs2007 (Vogtländer et al. 2010) and Stepwise2006 (Weidema 2009), both compatible with LCA software (SimaPro). After assessing and contrasting the obtained results, they conclude that it is not possible to state which LCIA method shall be preferred, as each method has its strength and drawbacks.

A more integrated approach is presented in Nafi et al. (2014), which introduces a method for the economic analysis of urban water management systems (UWMS) providing services, based on the principles of functional analysis (FA), Activity Based Costing (ABC) and Whole Life Costing (WLC). The cost structure is analysed according to the activities and physical flows comprising the primary and secondary functions of an UWMS. The method is not used in conjunction with an environmental assessment.

It appears that the economic sustainability of organisations and utilities comprising the urban water management systems is a real challenge, which has, to date, led to the development of a costing frameworks. From a policy perspective, the use of economic tools and principles for the achievements of the objectives of the Water Framework Directive (WFD, 2000/60/EC) is one of its most novel and interesting aspects. It is stated that all the costs assumed in the urban water cycle have to be recovered by the different agents involved. Cost-effectiveness analysis (CEA) has been widely adopted by organisations as the economic tool for the Programs of Measures (PoMs) of each river basin. In brief, CEA is a decision-support tool that enables the assessment of cost and effectiveness of different policy options (Martin-Ortega et al. 2012). In addition, multi-criteria decision analysis has been popular in policy-related works (Prato and Herath 2007), a tool which proved to be more appropriate for community-based approaches to water management. Recent literature (Termes-Rifé et al. 2013) though, suggests that Life Cycle Costing (LCC) could be a useful tool to calculate costs associated with the urban water cycle activities. Nevertheless, methodological improvements in its implementation are considered necessary to overcome controversial results.

2.6.2. Life Cycle Management and Resources

While life cycle thinking and assessment may play a critical role for more robust and comprehensive evaluation of resources, the existing life cycle methods are widely debated (Klingamir et al. 2014). Water and land use have to date been encountered as unique categories from LCA studies. The fundamental and various functions provided by water and its relevance to all areas of protection, place it apart from the other abiotic resources (Finnveden et al. 2009). Land use has been kept as its own category, since it is neither as clearly to be characterised in mass or volumetric terms, nor as biotic or abiotic (Goedkoop et al. 2009).

To address the gap of the traditional LCA view on water as an input flow, the United Nations Environmental Programme (UNEP) in collaboration with the Society of Environmental Toxicology and Chemistry (SETAC) Life Cycle Initiative working group launched the "WULCA" initiative (www.wulca-waterlca.org) (Koehler and Aoustin 2008) in August 2007. Their work has focussed on the assessment of water use from a life cycle perspective. Among the objectives of WULCA, is the establishment of adequate water inventories for LCA studies and the provision of guidance in the freshwater use modelling. The Water Footprint standard (BS ISO 14046, 2014) is the major outcome of the initiative and provides the principles, requirements and guidelines for the assessment of a 'water footprint' and for performing water-related LCA studies, which focus on supply-chain management, products, operations and inter-basin water "trade". A more detailed discussion to follow in section 2.6.3. and in Chapter 6.

Kounina et al. (2013) review the methods for addressing freshwater use in the life cycle inventory stage of a water-related LCA study. The review highlights that inventory methods generally suggest concepts for a systemic classification of freshwater elementary flows according to their type (surface, groundwater, precipitation water stored as soil moisture etc.) and describe technical water flows (e.g. irrigation water). Currently, there is a lively discussion from a growing body of academic literature (e.g. Hoekstra et al. 2016; Pfister et al. 2015; Ridoutt and Pfister 2013; van Hoof et al. 2013) on the advancements of the methods addressing water use from a life cycle perspective, through case studies or critical reviews.

For renewable resources it is less easy to draw the boundaries between the technical and the natural system; that is part of the explanation why it is difficult to describe effects of land use in LCA (Baumann and Tillman 2004). There are generally two categories of land use change: direct (i.e. modification of a land parcel) and indirect (effect of modified land use on other areas (i.e. and these have differing implications regionally and globally (Caffrey and Veal 2013). Despite the intrinsic link between water resources and land, an exhaustive literature review on the recent advancements and application of LCA in the arena of land use and its impact assessment is considered out of the scope of the undertaken research. A brief overview of the literature is presented, mainly focussing on LCA methodology and agricultural systems. In following chapters and when relevant to the needs of the research, literature will be retrieved to facilitate the discussion of the findings and their relevance to the wider picture of sustainability assessment of natural systems.

Agriculture is an incredibly diverse field. Variations in management practices exist in multiple scales, making it difficult for a general LCA to be conducted on agricultural activities (Caffrey and Veal 2013). The question of whether LCA can be applied to agricultural production systems was raised in 1990s (van der Welf et al. 2013). Since then, the rapid development in interdisciplinary research between agronomic, food/nutrition science and security and environmental systems analysis has boosted the parallel development of LCA methodologies in the broad agricultural sector. Several of its aspects are covered by recent LCA developments, ranging from general perspectives on LCA and food system sustainability (Soussana 2014; van der Welf et al. 2014), methodological improvements (Bello-Maurel et al. 2014; Pfister et al. 2014; Hospido et al. 2013) and case studies in agricultural and food production and consumption (e.g. Hörtenhuber et al. 2014; Lamastra et al. 2014; Xue et al. 2014; Ridoutt et al. 2014, Ruviano et al. 2014, Zonderland-Thomassen et al. 2014, Gerbens-Leenes et al. 2013, Herath et al. 2013, Milà i Canals 2010).

A major outstanding challenge in the application of LCA in agricultural system lies to the absence of a coherent approach to dealing with the issue of land use. Currently LCA methodologies use metrics of arable land use (m^2) to assess impacts, but more expanded definitions and boundaries are needed to assess specific impacts associated with land disturbance. An integrated approach assessing both environmental and economic aspects of land use has been adopted for the UK (Brandão et al. 2010), acknowledging soil management and fertilisation as the most dominant factors for climate impacts per monetary unit. Other works assess the impacts of land use on biodiversity loss (de Baan et al. 2013; de Souza et al. 2013) and on climate change (Perrin et al. 2014; Müller-Wenk et al. 2010). Methodological challenges and uncertainty related to data quality issues are also discussed across these studies. Geyer et al (2010) stress the necessity of modelling land use in a spatially explicit manner. Their work illustrates that a GIS-based inventory modelling of land use allows for important refinements in LCA theory and practice, while land use can be expressed as a set of elementary input flows.

The application of LCA and land use focus mainly on the assessment of agricultural systems and the issues arising relate to natural resources, land use change, livestock and management strategies. Other issues include economics, energy usage and societal concerns related to agriculture, as well as data requirements and uncertainty of the outcomes due to lack of reliable data. Future challenges may also consider the choice of the appropriate functional unit, the improvement of models for estimating emissions from biological factors, the understanding of systems' resilience and the transparency and presentation of the results (van der Werf e al. 2014). Similar needs and challenges are also identified in the application of LCA in site remediation services (Morais et al. 2010). The time-scale of the assessment, the importance of regional-specific modelling and the potential of LCA as a decision-making tool are highlighted as priorities. According to Soussana (2014), bridging the gap between LCA and natural capital assessment can be seen as key target for future research on the sustainability of agricultural systems.

Nevertheless, developing methods without considering application context should be avoided. On the contrary, publishing case studies and applications of LCA on production systems enhances interactions between scientific disciplines (van der Werf e al. 2014). Towards this direction, the integration of the concept of ecosystem services in the LCA framework has been proposed (Zhang et al. 2010), named as Eco-LCA. Recently, an ecosystem services approach has been applied on a case study for the Great Barrier Reef in Australia (Butler et al. 2011), analysing the trade-offs of different management scenarios on land use and water quality and exploring the potential use of this approach as a planning tool. This work provides a progressive step towards a generalised assessment at a catchment scale, while shedding light on the limitations of such an approach. In the European context, the development of the first atlas of ecosystem services at the scale of Europe (Maes et al. 2011) shows progress in the regionalisation of the ecosystem services prior of their inclusion in the LCA framework.

Together, the literature on the use of LCA in the water systems highlights current research interests and provides insights into future challenges. The growing number of case studies on the field suggest the importance of dealing with water challenges at a local scale, since the variables related to the natural –and thus to the urban-water cycle greatly vary spatially. By this time, focus in on arid or semi-arid regions, where scenarios have been assessed towards a sustainable water resources allocation. Moreover, in contrast to the focus of the majority of LCA case studies, these works also address the impacts related to water use, in terms of both quality and quantity requirements. In addition, the need to move from the plant scale to a broader approach is highlighted in many studies. System boundaries have expanded to include the whole urban water cycle and in few case a city or a river basin. Whilst the boundaries are enlarged, the advancement of characterisation factors to address problems at regional scale is becoming more important. In the same vein, many works stress the improvement of data quality and the reduction of uncertainty in the results. Last but far from least, enhancement of stakeholders' participation and links of LCA to costing and societal aspects to complete the whole picture of sustainability are thought to be necessary. Methodological improvements for the linkages between LCA, LCC and evaluation methods are required.

In summary, the LCA frameworks related to natural resources have shown progress over the last few years, but there are still many challenges to be addressed. While a number of methods addressing freshwater and land use have emerged, more applications and case studies will reveal their applicability, strengths and weaknesses. The complexity of natural systems entails an integrated and holistic approach towards their assessment; thus, the application of Life Cycle Thinking could be tested as part of a sustainability assessment.

2.6.3. Water Inventories and Water Footprints

Life Cycle Thinking has a dynamic presence in the field of Water Accounting, mainly with the form of Water Inventories. Literature shows a rapid development of Water Inventories as part of the well-established methodology of LCA, which focus on the creation of indicators that can best describe the water use or consumption over the lifespan of a product (system, process etc.) and the water-related environmental impacts.

In the arena of Water Inventories, there is a parallel development: the Water Footprint Assessment (WFA) as described in its standardised version in the Water Footprint Network manual (Hoekstra, 2011) and the Water Footprint ISO standard (henceforth LCA_{water}) (BS ISO 14046:2014) which is developed as a means to improve the assessment of water-related impacts within LCA studies. Both methodologies aim to help practitioners preserve water resources. However, their approach, focus and level of applicability differ in these research streams. Boulay et al. (2013) compare the two methodologies, summarising their similarities and differences (**Figure 2.7**): both methodologies comprise of four steps (goal & scope definition, accounting/inventory phase, impact assessment, response/interpretation), but the use of quantitative indicators is differentiated (accounting versus impact assessment phase). The article concludes that the methodologies are fulfilling complementary goals and their synergetic use could benefit future progress. It appears that both developments also share a number of methodological limitations, mainly related to the lack of standardised approaches for the quantification of water amounts and to the resources and data intensity of their application.

Many researchers and practitioners are puzzled by the different types of assessments and interpretations of the WFA's method and the draft ISO's norm, particularly by the differences in the definition of "water footprint", while the publication of multiple standards leads to further confusion (Tillotson et al. 2014).

In the ISO standard, the 'Water Footprint' is defined as a metric that quantifies the potential environmental impacts related to water, while the accounting phase of a water-related LCA is conducted as part of the inventory phase (LCI). From an LCA perspective, the Water Footprint Inventory is the compilation and quantification of inputs and outputs related to unit processes that make up the product system. It does not include merely water volumes, but all inputs and outputs of a product system that may result in environmental impacts associated to water as a resource. A water footprint can be represented as a result of a stand-alone assessment or as a sub-set of results of a larger environmental assessment, such as a full-LCA.

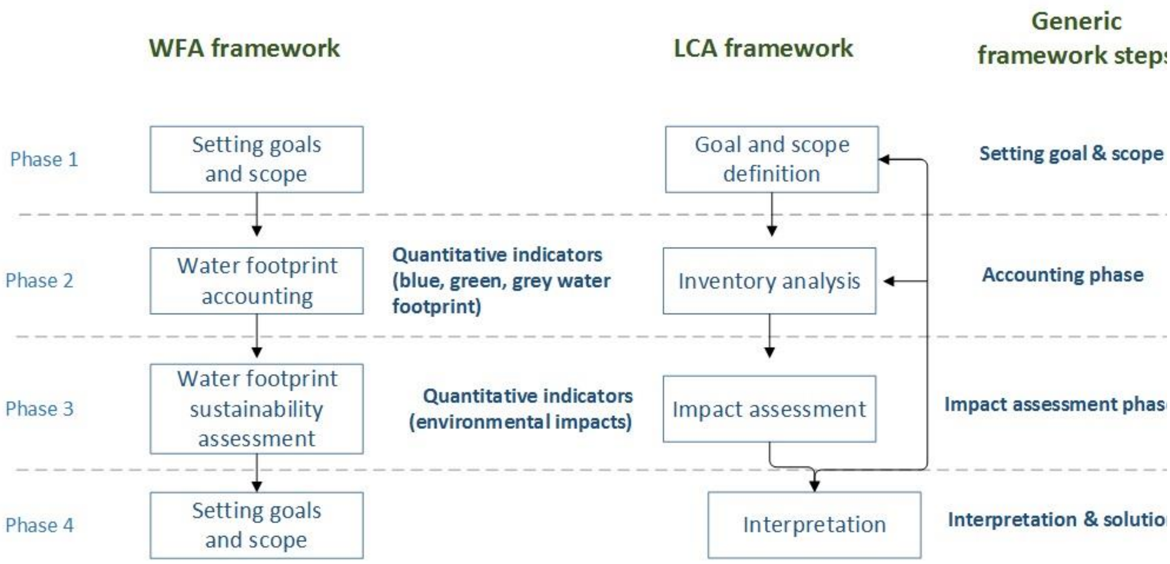


Figure 2.7.: The comparison between the Water Footprint Assessment and the water Life Cycle Assessment frameworks –adapted from Boulay et al. (2013). Both methodologies comprise of 4 steps: goal & scope definition, accounting/inventory phase, impact assessment, response / interpretation. The use of quantitative indicators is differentiated (accounting versus impact assessment phase).

As originally defined by the WFA working group, the water footprint (WF) is an indicator of freshwater use that looks at both the direct and indirect use of water of a consumer or a producer (Hoekstra 2003). It is a volumetric, multidimensional indicator, showing water consumption volumes by source and polluted volumes by type of pollution. The WF can be calculated for different entities (e.g. a step, a process, a product, a nation etc.), different groups of consumers (e.g. an individual or a family) or producers (e.g. an enterprise or an economic sector) and for different geographically delineated spatial scales (e.g. a country, a region or a catchment). The WF is a geographically and temporally explicit indicator, showing not only the volume of the consumptive water use and pollution, but also the locations and time. It is not, though, an indicator of severity of the local environmental impact of water consumption and pollution; therefore, it does not address environmental issues other than freshwater scarcity and pollution.

The augmented number of concepts and methods related to the assessment of freshwater use, has driven the publication of a number of review papers (Núñez et al. 2016; Boulay et al. 2015a,b; Kounina et al. 2013; Berger et al. 2010) and case studies (Boulay et al. 2015 a,b,c,; van Hoof et al. 2013; Godskesen et al. 2013; Yang et al. 2013; Angrill et al. 2012; Gleeson et al. 2012; Stoeglehner et al. 2011; Pfister et al. 2009), which address regional water resources at various scales (product, aquifer, hinterland, urban, groundwater catchment, watershed).

Literature shows a dynamic progress in the methodologies addressing water-related impacts in LCA studies (e.g. Bayart et al. submitted) which will be extensively discussed in a following chapter (Chapter 6). The advancements are also highly debated (Hoekstra 2016). The outstanding challenges mainly relating to the data quality and management in water-related LCA studies are addressed in recent literature. Pfister et al. (2015) described the improved version of the Ecoinvent database (Ecoinvent version 3.1) The advancements allow the inclusion of relevant flows to address water use in LCA and calculate WF on the product level for most processes, including uncertainty information. The comprehensive data collection of water use data is at the process level, facilitates the assessment of water use within LCA and water footprinting beyond agricultural production and enhances the transparency in the calculations. Nevertheless, data quality and spatial resolution issues still remain. From a scientific point of view, a high spatial resolution is preferable for the inventory and impact assessment phases (Pfister et al. 2011), while practitioners are often satisfied with country-level resolution (Vionnet et al. 2012).

The concept of WF and the WFA methodology have been broadly accepted by global and national policy-makers and substantially influenced strategic planning at a regional scale. The outputs of these projects have highlighted areas of improvement in different research areas and economic sectors or even formulated responses for specific regions. In the research arena, the WFA methodology has been widely applied. A recent Scopus search showed that during the last five years (2012-2016), over a 100 research papers have been published following the WFA methodology for diverse topics, ranging from methodological improvements to case studies and future scenario analyses. Although it has been mainly applied in agricultural systems and products (e.g. Ran et al. 2016; Hess et al. 2015), its application at the urban water cycle level is showing significant progress (Manzardo et al. 2016). A more detailed methodological discussion follows in Chapter 6.

In the published works, there is a unanimous agreement that the development of WF has driven substantial progress in the elaboration of water use in the production and consumption of final products at different geographical scales and in the quantification of water regimes for complex environmental and economic systems. Nevertheless, the WFA methodology has received severe criticism (Pfister et al. 2016; Wichelns 2015; Chenoweth et al. 2014; Yang et al. 2013) mainly in regards to the limitations and shortcomings with regard to policy relevance, data accuracy, methodological approaches and conceptual consistency. The value of WF for policy-making in water resources management is considered unclear or limited (Chenoweth et al. 2014; Perry 2014, Yang et al. 2013). The criticism focusses on the limitation of the WF to shape optimal strategies, mainly regarding issues such as water scarcity or international trade (Wichelns 2015). It is also criticised for not being an analogous to other existing environmental footprints, and for its seemingly simple and misleading single production unit and form. According to Yang et al (2013), more studies at different scales are required, along with the adoption of interdisciplinary approaches to allow WFA to include issues relating to climate change and uncertainty and to harmonise the conceptual bases of the components of the WF.

2.7. Overview of the literature: research gaps and opportunities arising

The literature shows a consensus that, currently, the priority areas in the field of water science and sustainability include the adoption of a systemic approach to water challenges and the creation of systemic, integrated approaches to address sustainability issues. It is highlighted that such approaches would have value both for academic research and industrial applications, especially in the field of strategic asset management. They would also enable the implementation of national and international policy demands, while pulling together the fragmented field of sustainability.

The essence of examining the “local context” when assessing water impacts has also emerged, relating to the discussion that water challenges are strongly dependant on the local factors at a catchment scale, such as the ecosystem, communities or water users. This is much relevant for the water sector, which is encouraged to adopt systemic approaches for the management of their asset systems. The practice to date reveals a rather introverted approach to asset management strategies with infrastructure (i.e. built and financial capital) at their core. However, the more holistic and systemic examinations of the local context and the identification of water-related “hotspots” would assist companies to prioritise investment or risk-mitigating actions, such as policy engagement and community outreach. The creation of approaches that would enable businesses to integrate natural capital in their planning and practice has been recognised as a priority area for future research (Natural Capital Initiative 2015).

The advancements of water inventorying and impact assessment methods have, to date, enabled studies at an urban level with focus on the urban/artificial water cycle. Thus, the “local” aspect has been addressed, but not within a geographically delineated boundary, such as that of a catchment (or watershed). The latter would enable comparisons among well-defined systems, the shift towards uniform assessments, and the adoption of a holistic view, more relevant to policy demands. Enlarging the focus of asset management to wider systems would enable the integration of local and diverse elements- such as the ecosystem and its services- in the asset management portfolio of the water sector.

Table 2.3.: The field of Water Accounting. Water Account are based on the National Statistic Accounts and have expanded to include physical flows. Water Inventories are mainly related to the concept of ‘Water Footprint’, where two parallel developments are observed: water-related LCA (LCA_{water}) and Water Footprint Assessment (WFA).

Water Accounting	
Water Accounts	**Water Inventories (& Footprints)**
Accounts in tabular format	Water indicators and metrics
Based on macro-economics & the System of National Accounts	Based on Life Cycle & Water Footprint Assessments
Sector level & National scale	Product level & Multiple scales
Inform sectorial & governmental decision-making	Inform sectorial decision-making & practitioners
Direct policy relevance (WFD, Phase 2 of RBMPs)	Indirect policy relevance
Data quality & availability limitations	Data quality & availability limitations
Multiple accounts (e.g. SEEAW, AWAS) & outstanding methodological challenges	Multiple indicators, ambiguity in terminology & outstanding methodological challenges

The application of Life Cycle Thinking through the joint use of the Life Cycle Management tools, has been identified as a prominent framework towards achieving a robust assessment of sustainability issues, the design of effective strategies and the formulation of well-informed decisions. In the arena of water research, the advancements in the field of Water Accounting are promising for the performance of sustainability assessments of water systems.

Literature shows evidence of the parallel development of two streams of research for Water Accounting (**Table 2.3.**). Water Accounts provide a structured, tabular format which is based on the System of National Accounts. Their methodology has been altered to include the accounting of physical flows, such as water, at a sectorial or national level. Their advancements have the potential to inform sectorial and governmental decision-making, as they have direct relevance on current policy demands. There is a growing volume of academic literature on the advancements of Water Inventories with the development of multiple indicators under the umbrella of the concept of 'Water Footprint' and two types of environmental assessment: water-related LCA (LCA_{water}) and Water Footprint Assessment (WFA). Their methodologies are constantly evolving, resulting in a lively research dialogue. Their advancements to date allow for environmental assessment of freshwater use mainly at a product level, but for multiple scales. Their indicators and results have the potential to indirectly inform decision-making at sectorial or industrial level.

Despite the advancements, the field of Water Accounting suffers from a lack of harmonisation among the available methodologies, while further methodological improvements are also required. The quality and availability of data necessary for the performance of the assessments or the construction of the accounts hinder their systematic use.

The two prominent methodologies form the field of Water Inventories use the term of 'water footprint' (WF) to describe different aspects of the environmental assessment, causing terminological ambiguity. Thus, WF is a volumetric indicator for the WFA methodology, but an impact-related figure for LCA_{water}. In the WFA literature, the concept of WF as a volumetric indicator has been widely used in agriculture and more recently in urban water systems, but the impact assessment phase of the methodology is rather immature and relevant only to water assessments. On the other hand, LCA is a well-established tool and water-related LCA is flourishing, especially in regards with the development of impact indicators and the standardisation of cause-effect relations for water pathways. The issues arising from these parallel and rather competitive research developments are related not only to the terminological confusion, but also to the lack of harmonisation and the unavailability of adequate data at different levels and scales. Further discussion and more practical applications are needed to show the real value of the recent developments and for a consensual agreement on the tools applied.

In the arena of LCA, a growing volume of literature suggests the expansion of system boundaries for water studies, in order to include the water cycles as wholes. The impact assessment needs to address water use, in terms of both quality and quantity requirements, at a regional scale. Whilst the boundaries are enlarged, the advancement of characterisation factors to address problems at regional scale is becoming more important (Arbault et al. 2014). In the same vein, many works stress the improvement of data quality and the reduction of uncertainty in the results. Last but far from least, enhancement of stakeholders' participation and links of LCA to costing and societal aspects to complete the whole picture of sustainability are thought to be necessary. Methodological improvements for the linkages between LCA, LCC and evaluation methods are required. As pointed by McManus and Taylor (2015) for the further development of the LCA methodology, "a consistent approach is required across sectors, which starts with uniformity in systems boundaries".

Together, literature shows a unanimous need for integrated, systemic approaches at a regional scale. Despite the research developments in the field of natural resources accounting and environmental assessment, many methodological challenges are still to be addressed. While a number of methods addressing freshwater and land use have emerged, more applications and case studies will reveal their applicability, strengths and weaknesses.

The complexity of natural systems entails an integrated and holistic approach towards their assessment, while the localised character of water resources stresses the need for shifting towards regional assessments. Thus, the application of Life Cycle Thinking could be tested as part of a sustainability assessment at a catchment scale.

Chapter 3: Research Framework

Chapter 3 focuses on the approach of the research and presents the aim and objectives, along with the methods and methodology adopted to conduct the research. Throughout, specific choices are made to serve both research and pragmatic purposes; i.e. industrial requirements, related to the strategic planning of the industrial partner of the doctoral research project (i.e. Wessex Water Services Ltd). Thus, it can be categorised under action research, in terms of its practical nature. This type of research aims at dealing with real-world problems and issues (Denscombe 2010). However, as action research is quite clearly a strategy, rather than a research approach, in this work it serves as the link between research and practice, when it comes to the selection of the location (i.e. catchment) under study.

The work cuts through various disciplines that range from asset to water management, but also include environmental and catchment science, ecosystem services, policy-making, and, finally, systems thinking. The "multi-disciplinarity" of the work, in combination with its aspiration to address real-world issues through a holistic approach, classifies the research under transdisciplinary research.

3.1. Transdisciplinary Research

According to Leavy (2011), transdisciplinarity is a social justice oriented approach to research in which resources and expertise from multiple disciplines and stakeholders (academia, industry, policy) are integrated in order to holistically address a real-world problem. It is issue- or problem-generated, not discipline-driven (Krimsky 2000), and thus, it is a way of putting the research problem, topic, issue or question at the centre of research process, irrespective of one's "home" discipline. The research questions are framed according to the real-world problems that need to be solved and based on the sets of disciplinary knowledge necessary at each stage. Transdisciplinary approaches are increasingly encouraged as they are more likely to make research more useful to a range of stakeholders (academics, policy makers, the society) and enable the research undertaken to broadly reflect the interests of those involved (Bracken et al. 2015).

Many researchers suggest that transdisciplinarity is not a method for doing research or an outcome of research, but rather an approach to the research process (Klein 2004; Lawrence and Després 2004) or a "new way of thinking" (Giri 2002). It is a goal-oriented process rather than a knowledge production process per se (Walter et al. 2007), which enables researchers to transcend disciplinary limitations and create new knowledge through the combination of theories, methodologies and data.

Transdisciplinarity involves the adoption of a systems view, which is a powerful concept to complex research projects (Schwaninger et al. 2007). Its overall goal is to provide a holistic and synergetic approach to studying the issue or problem and to enable researchers to build conceptual and methodological frameworks (Leavy 2011). The key principles of transdisciplinary research also include transcendence, emergence, synthesis, integration, innovation and flexibility (Leavy 2011; Lawrence 2004) (**Table 3.1.**).

Although multi-disciplinarily and inter-disciplinarity have formed the basis for the development of transdisciplinary research, these terms differ significantly on the degree of integration of concepts, theories, methods and findings involved (**Table 3.2.**), but also on the level of interactions among disciplines and collaboration among researchers (Cameron and Mengler 2009).

Table 3.1.: Principles of Transidsciplinarity, according to Leavy (2011).

Principle	Practice
Issue- or Problem- Centred	Problem at centre of research; determines use of discipline & resources and guides methodology
Holistic or Synergetic Research Approach	Problem considered holistically through an iterative research process which produces integrated knowledge
Transcendence	Researchers build conceptual frameworks that transcend discipline perspectives in order to effectively address the research question
Emergence	Placing the problem at the centre of research cultivates the emergence of new conceptual and methodological frameworks
Innovation	Researcher build new conceptual, methodological and theoretical frameworks as needed
Flexibility	Iterative research process requires openness to new ideas and willingness to adapt to new insights

Table 3.2.: A comparison of Multi-Disciplinarity, Interdisciplinarity and Transdisciplinarity, according to Leavy (2011).

	Level of Collaboration between Disciplines
Multi-Disciplinarity	Collaboration between two or more disciplines without integration
Interdisciplinarity	Collaboration between two or more disciplines with varying levels of integration of concepts, theories, methods, findings
Transdisciplinarity	Collaboration between two or more disciplines with high levels of integration causing the development of new conceptual, theoretical and methodological frameworks

Evaluation is a particularly thorny issue in transdisciplinary research because no clear peer community has yet been firmly established (Stavridou and Afonso 2010; Wickon et al. 2006). Nevertheless, transdisciplinary research can largely be evaluated with respect to effectively addressing the issue or problem at hand, its focus on the research objectives, the use of appropriate strategies, and largely, against its key principles.

The research undertaken is transdisciplinary in nature, which will be evaluated in a later stage (Chapter 7), against the criteria described above. The transdisciplinary approach selected for the research project is reflected in the methodological choices made throughout, from the selection of the tools and techniques, through to the critical analysis of the outputs in terms of their suitability to provide a holistic view to water and asset management at a catchment scale.

3.2. Research Aim & Objectives

The **aim** of the research is to provide a catchment scale modelling schema for holistic asset management in the water sector.

In order to achieve the aim, the following research objectives have been identified:

1. Define holistic asset management.

2. Select the techniques and define the rules for the creation of the catchment scale modelling schema.

3. Determine the tools and create the rules for the assessment of the environmental performance of holistic asset management strategies.

4. Investigate the applicability of the modelling schema and the environmental performance assessment through an industrial case study.

5. Evaluate the practical value of the research outcomes through a critical analysis.

3.3. Research methods

Transdisciplinary research projects typically require the use of more than one methods (i.e. tools used to gather and interpret data), which are selected for the utility to serve the specific problem or issue under study. Therefore, transdisciplinary projects often involve multi-method or mixed-methods designs which are constructed in service of the research goals.

Mixed-methods provide a practical approach to research problems, emphasising on pragmatism. After Hesse-Biber and Leavy (2011), mixed methods and multi-methods designs –in their best form- offer holistic approaches to research, where each component of the research speaks to other components. In other words, in their best execution, the use of multiple methods is not simply additive, but rather, the use of each method informs the use of the other methods.

The decision of tools and techniques within a mixed-methods strategy is based on how well they fit within the research philosophy (Leavy 2011; Denscombe 2010). This approach is 'problem-driven' in the sense that it treats the research problem as the overriding concern and adopts a pragmatic position that allows to bring together methods drawn from 'paradigms' of research conventionally regarded as incompatible (Denscombe 2010). Moreover, it allows the simultaneous use of both quantitative (QUAN) and qualitative (QUAL) tools within a single research project, while focusing on the need to explain why the various approaches are beneficial and how the alternatives are to be brought together.

In the frame of the doctoral research, the mixed methods approach has proven beneficial in several stages and for different purposes. At first, the creation of the modelling approach and schema, as described in the research aim (section 3.2.), has been based on the conjunction of various tools –both quantitative and qualitative- which have been selected as the most appropriate to fulfil the scope of the work. In addition, the creation of the methodology for the assessment of the environmental performance of the catchment-based strategies has been based on standardised methods- such as Life Cycle Assessment (ISO 14046:2014; ISO 14040:2006)-, which involve both quantitative (e.g. inventory and impact assessment) and qualitative (e.g. interpretation) stages. Further, the computation of environmental outputs has been conducted with mathematical forms and indexes, while the use of specific software has supported the computation of several features. The data analysis has been performed according to scientific methods and the discussion of the results and research outputs is based on an extensive literature search. Detailed description of the methodological choices made will be extensively discussed in the chapters discussing the creation of the modelling schema and of the environmental performance assessment (Chapters 5 and 6 respectively).

In terms of data collection methods, documents (reports, literature), datasets and databases have been used. Since a part of the research focuses on a particular case study, preliminary data is essential to better describe the current status of the selected catchment and perform the environmental assessment. However, in order to serve the scope of the research within the limited time assigned, it has proven more pragmatic to rely on secondary data, while rigorously evaluating the credibility of their sources. The industrial partner of the research has provided catchment-specific reports and data. In addition, datasets on climatic and environmental parameters have been accessed when appropriate. When required and if applicable, published data and results from relevant studies across the UK or the globe were used. Data collection was predominantly driven from the needs of the methodological choices, as will be discussed in a later stage (Chapter 6).

3.4. Rules for formulating a methodology for transdisciplinary projects

The design of transdisciplinary research requires an evolving approach that follows an iterative or responsive process where the methodology matures over the course of the research process as a result of new learning (Wickson et al. 2006). A responsive approach to research design helps to ensure that the research problems and questions stay at the centre of the research process. Moreover, the research design strategy should be holistic and involve a synergetic approach to research (Leavy 2011).

While transidsciplinarity allows to research complex problems, researchers may select topics in a variety of ways, including their awareness of a pressing need/problem/issue in the society. Transdisciplinary research topics may be organised around a “site”, i.e. a conceptual space where disciplines assemble (Krimsky 2000). Thus, situational context becomes important in studies of the concrete real world, whose results need to be comparable or transferable. This is a strong argument for a case study method in transdisciplinary research (Walter et al. 2007). A case has to be selected from the viewpoint of science, which aims at deriving generally valid insights (Yin 2009). Therefore, a case represents a general problem, but in a specific and unique shape. The question of the transferability of the results has to be part of the scientific research (Walter et al. 2007).

The case study method, as defined by Yin (2009) is an empirical inquiry that investigates a contemporary phenomenon within its real-life context, addresses a situation in which the boundaries between phenomenon and context are not clearly evident and use multiple sources of evidence. The case study is a method of choice when the phenomenon under study is not readily distinguishable for its context (Yin 1993). According to Denscombe (2010), the case study approach can use a wide range of phenomena as the unit of analysis, but, in order to qualify as something that lends itself to case study research, it is crucial that the unit has distinct boundaries. In the case of a catchment-based approach, the boundaries are those of the watershed, as defined by its geophysical and geographical structure.

The aim of case studies is to illustrate the general by looking at the particular. The logic behind concentrating efforts on a single case rather than many is that there may be insights to be gained from looking at the individual case that can have wider implications and, importantly, that would not have come to light through the use of a research strategy that would cover a large number of instances (Denscombe 2010). Many of the features associated with a case study are not necessarily unique; however, when combined, they give the approach its unique character. The rigor of a case study should be judged by validity and reliability (Yin 1993).

Further to the well-known use of case studies to develop new hypotheses, the case study method can serve evaluation needs by being able to assess outcomes and test hypotheses (Yin 1993). To achieve that, a major prerequisite is the development of causal relationships, which will then become the main vehicle for developing generalisations.

Life Cycle Management tools (e.g. LCA) could prove useful for transdisciplinary research, provided their iterative character in the sustainability assessment of impacts of a product/system/activity using both quantitative qualitative tools. Case studies can illustrate how effective life cycle management approaches -such as the combination of LCA with other techniques- could become in practice, when used to evaluate sustainable alternatives for product systems (Klöpffer et al. 2008).

3.5. Research approach

The research undertaken explores how life cycle management tools and their underpinning rationale can inform the creation of new modelling methodologies, applied to catchment systems. Based on the principles of transdisciplinary research, a flexible approach has been followed for the creation of the modelling schema. The selection of methods from other disciplines have been employed to overcome methodological barriers imposed by the life cycle management tools. This has enabled not only the creation of a transdisciplinary modelling methodology, but also knowledge transfer across disciplines, such as hydrology and integrated catchment management. The modelling methodology includes two main features: the modelling schema, which represents the model's external structure, and the modelling inventory, which represents the internal anatomy of the model.

The research outputs were then implemented in an industrial case study. A rural catchment system has been selected as the case study, whose example has revealed the strengths and limitations of the methodology created. The contribution of the methodology to current practice and needs of the water sector is then discussed, especially in regards to policy compliance and strategic asset management planning. The discussion evaluates the transdisciplinary nature of the research and highlights areas of future work, especially for the further development of the research outputs, their reproducibility to other catchment systems and their contribution to other research areas.

A modular research approach has been followed throughout the research (**Figure 3.1.**). For each of the research objectives – as defined in section 3.2.- an individual research approach was selected. The structure of this document reflects the modularity of the approach. Thus, the main research outputs are presented in Chapters 5 and 6, while the critical evaluation of the methodologies and of their practical value are discussed in Chapter 7. The steps undertaken to meet the individual research objectives have contributed to meet the overall aim of the research.

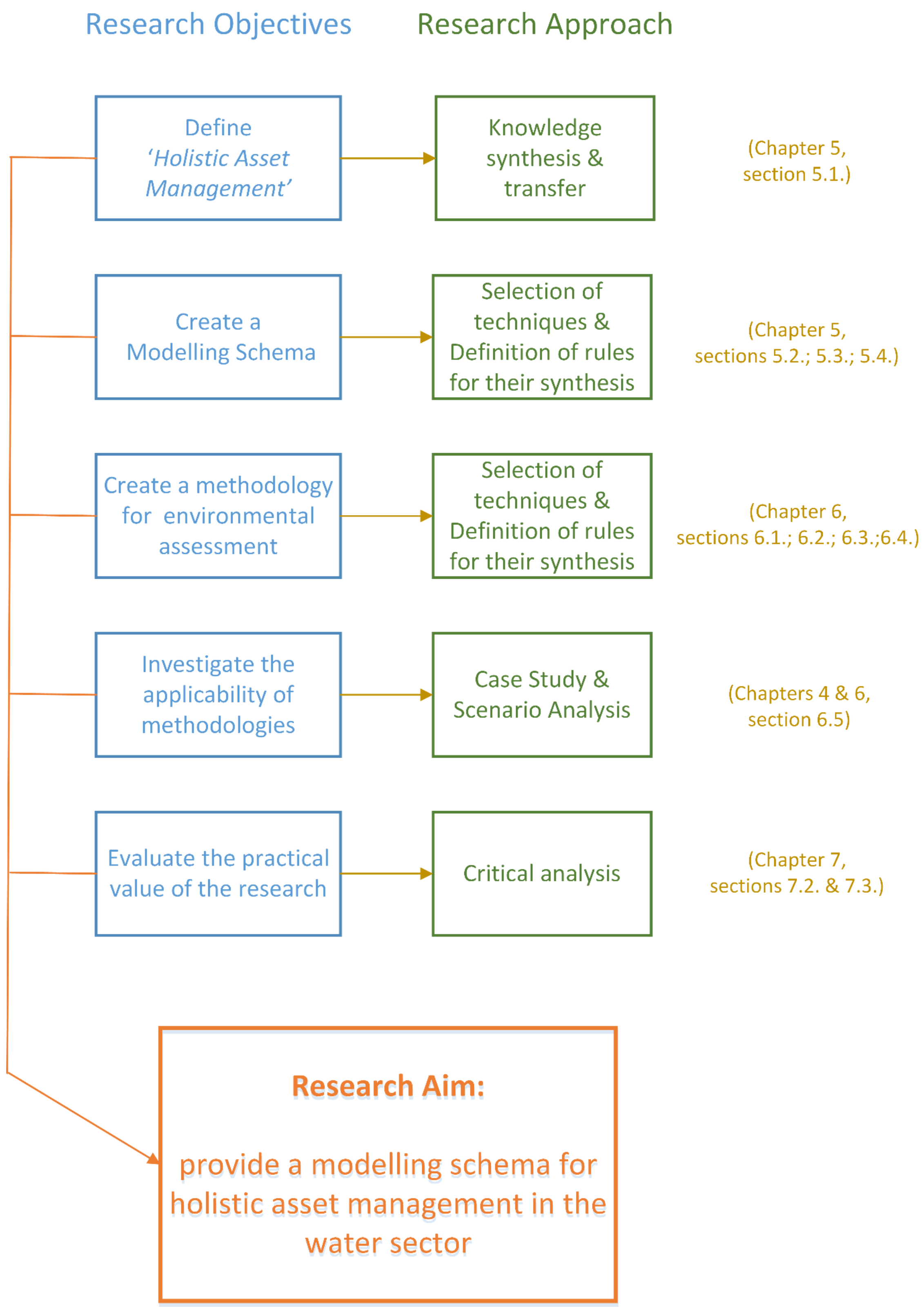


Figure 3.1.: Modular Research Approach. For each of the research objectives, an individual research approach was selected. The steps undertaken contribute to the completion of the overall research aim.

Chapter 4: Case Study

The chapter introduces the industrial partner of the research project and discusses their strategies and planning for asset and catchment management. Then, the catchment selected as a case study for the needs of the research is presented, along with an overview of its key environmental issues. Based on the findings of the latter section, the final part of the chapter focusses on the studies undertaken by third parties for the same catchment system and analyses their recommendations for tackling key issues in the catchment.

4.1. Industrial Partner: Wessex Water Services Ltd

Wessex Water Services Ltd (henceforth referred as: WSX), an YTL Power International company, is the industrial partner of the research project. They are a regional water and sewerage business serving 2.7 million customers across the south west of England (**Figure 4.1.**) including the areas of Dorset, Somerset, Bristol, most of Wiltshire and parts of Gloucestershire and Hampshire. Among the primary aim of WSX is to secure excellent standard of service by providing high quality water and environmental services that protect health, improve the environment and give customers good value for money. The efforts and continuous improvements have been rewarded by the economic water industry regulator, Ofwat, who has recognised WSX as one of the most efficient water and sewerage companies in England and Wales.

As analysed in their latest public reports (Wessex Water Services 2014), WSX investment strategies have managed to transform their customer service, to increase the company's efficiency and to achieve major improvements in the water environment by securing compliance with environmental standards. To this end, WSX has delivered a substantial environmental programme, which is driven by European regulations and overseen by the environmental regulator, the Environment Agency. As a means of addressing local issues, WSX has trialled the adoption of catchment management strategies for tackling pollution levels, instead of implementing typical end-of-pipe solutions. To support such approaches, they have created a team of farm advisers who work with farmers and landowners and provide practical, evidence-based help and advice on methods of land management that protect water quality. In some cases, they have also provided financial incentives to farmers and established collaborative actions with other organisations (e.g. Poole Harbour Catchment Initiative).

Figure 4.1.: Wessex Water's region of service. Map abstracted from the publicly available reports of Wessex Water Services Limited.

The latest business plan of WSX highlighted nine major outcomes to be addressed from 2015-2020 and beyond (**Figure 4.2.**). In terms of their environmental strategies, the sustainable and efficient implementation of the Water Framework Directive (WFD, EC 2000/60) is prioritised. As a broad principle, WSX aims to avoid the end-of-pipe solutions as their costs are becoming increasingly disproportionate to their benefits for the environment. Instead, the role of catchment management is promoted in the business plan, as a means to achieve an innovative, low-carbon programme which can tackle the causes of problems rather than just relieving the symptoms. In regards to catchment strategies, WSX plans to extend the incentives and adopt integrated solutions that improve the water quality at source, protect water resources and enable stakeholders to participate in the implementation process and its beneficiary outcomes. To establish these innovative approaches as the norm, a greater level of transparency and partnership between the regulators, the regulated industry and other external stakeholders is required (Wessex Water Services 2014).

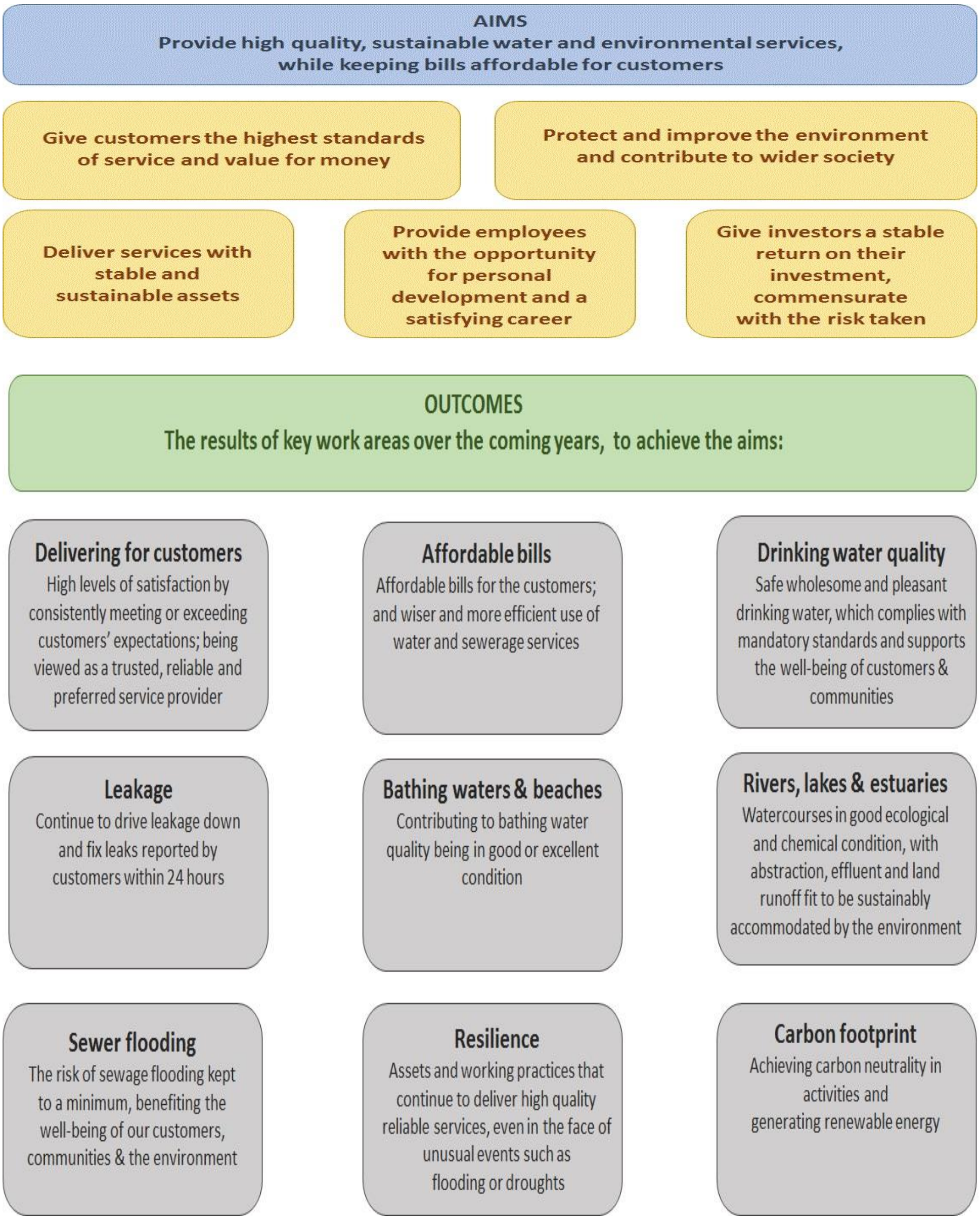


Figure 4.2.: Wessex Water aims and specific outcomes for the period 2015-2020 and beyond, as described in their current business plan.

4.2. Case Study: The Poole Harbour Catchment

An example catchment has been selected for the needs of the undertaken research based on both industrial and academic criteria. The Poole Harbour Catchment is among the first pilot locations where the catchment-based approach was trialled. As a result, a plethora of comprehensive studies were undertaken creating a considerable amount of secondary data available for further use. The results of the studies were summarised in the Nitrogen Reduction Strategy (NRS) report, published by Natural England, in collaboration with the Environment Agency and Wessex Water. The catchment's area and setting are introduced in this section, along with an overview of the projects and actions undertaken to date by third parties.

4.2.1. Study Area

The Poole Harbour Catchment is located in South-West England, in the county of Dorset **(Figure 4.3.)**. The catchment area is predominantly rural in character. Other than the conurbation of Poole and Bournemouth that extends outward from the north of the harbour and across the watershed, the only major settlements are Dorchester and Wareham. The total population is about 210,000 (2011 data).

Inflowing rivers in the harbour cover a drainage area of about 820 Km^2. The substantial part of the catchment lies to the west and is drained by the River Frome and the smaller River Piddle. To the north and south are the much smaller catchment areas of the Sherford River and Corfe River respectively, and also the catchments of several minor streams.

Poole Harbour occupies a shallow basin at the confluence of several rivers and streams which flooded as a result of rising sea level. Poole Harbour has a distinct lagoonal character and is one of the largest and shallowest natural harbours in the world; with an area of approximately 38. Covering an area of about 3,300 ha, the Poole Harbour accounts for about a quarter of the saline lagoon habitat in England and Wales (Langston et al. 2003).

The area contains many sites of local, regional, national and international importance, with a range of habitats supporting a variety of species. Poole Harbour and its surrounding wetlands were first designated as a Site of Special Scientific Interest (SSSI) in 1964 and the site has been periodically revised, most recently in 1990. The special interest of the harbour itself lies in the estuarine habitats and in particular species and assemblages of species these habitats support. In 1998 the harbour was designated both as a Special Protection Area (SPA- European site) and as a Ramsar site (Ramsar Convention 1971). A substantial proportion of the area is within the Dorset

Area of Outstanding Natural Beauty (AONB), while a large part of the catchment is also within the Wild Purbeck Nature Improvement Area (NIA) (**Figure 4.4).**

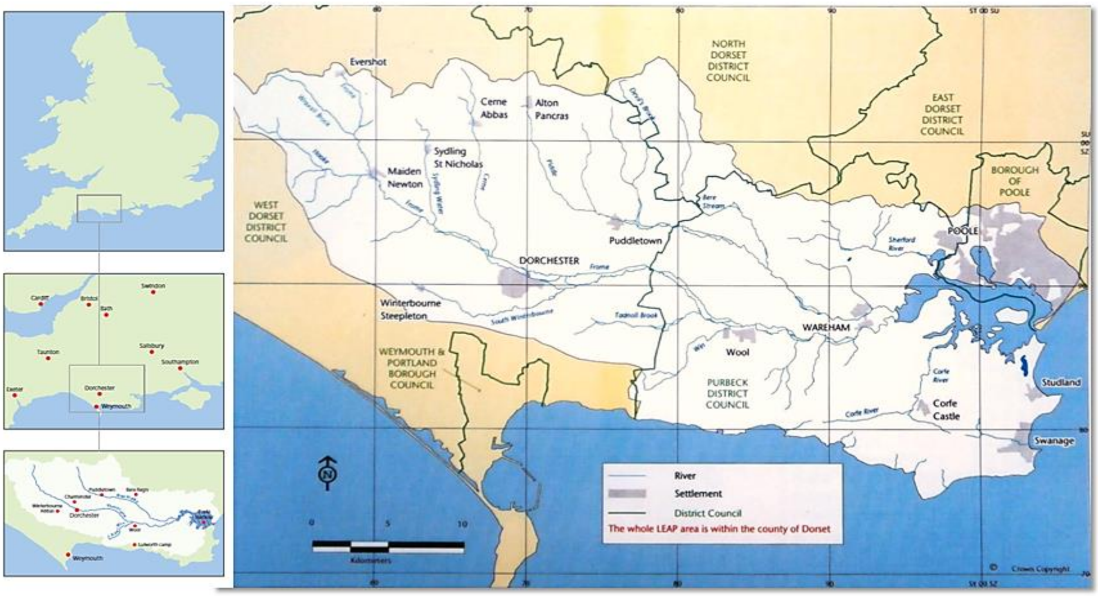


Figure 4.3.: The location of the Poole Harbour Catchment and its major settlements.

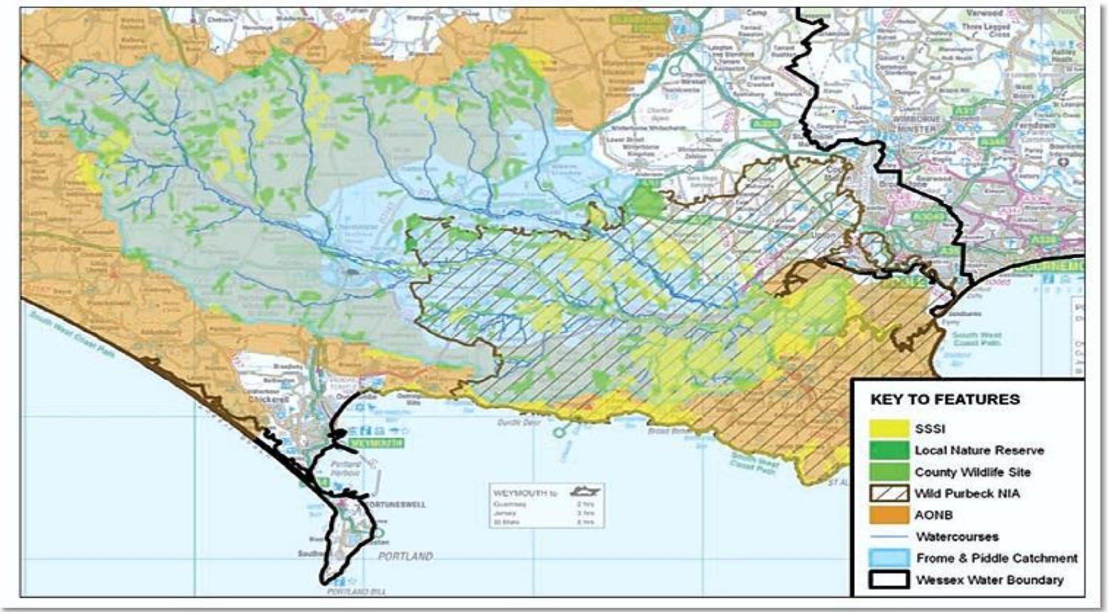


Figure 4.4.: Preserved sites within the Poole Harbour Catchment, protected under national or European conventions.

Poole Harbour is also designated as 'Protected Area' under the European Water Framework Directive (WFD, 2000/60/EC). It is classified as a heavily modified transitional waterbody under the WFD (Waterbody ID GB520804415800), because of its modification for coastal protection and navigational purposes, which flows in the Dorset-Hampshire coastal waterbody (**Figure 4.5.**). The harbour is highly eutrophic and there is a very clear disturbance from the excess of nutrients in this system. There are requirements under the Directive for the harbour to achieve Good Ecological Potential (as a heavily modified waterbody) and to meet the standards and objectives of the European Protected Area designations. As the Dorset-Hampshire coastal waterbody is also currently failing for nitrogen, it is believed that potential improvements made to upstream waterbodies (Poole Harbour) could result in an improvement of the status of this larger coastal waterbody.

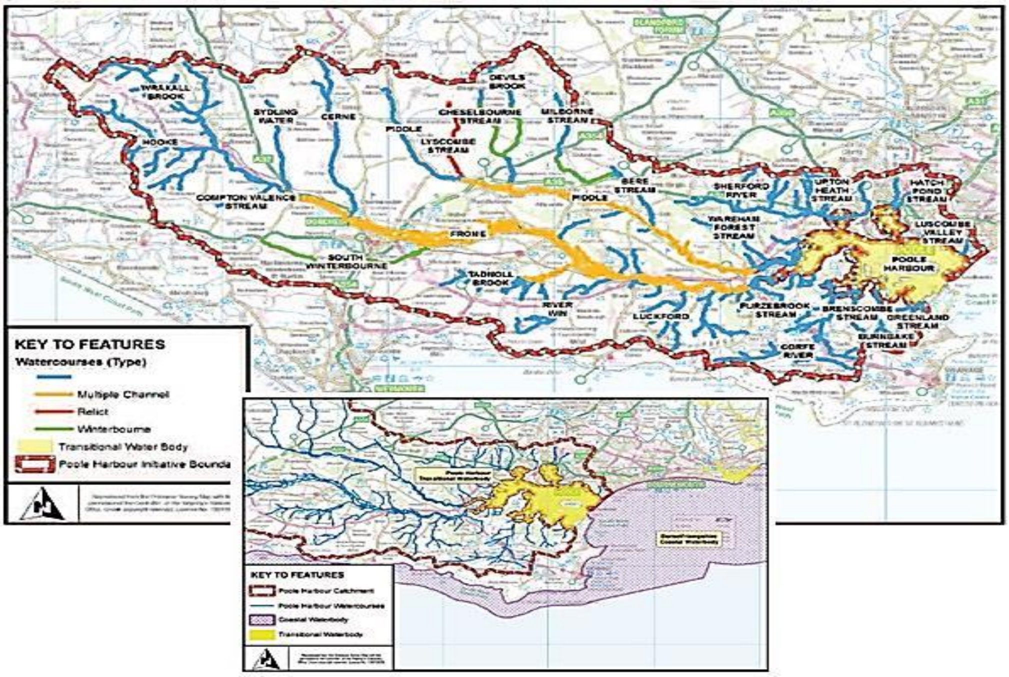


Figure 4.5.: Poole Harbour Catchment waterbodies according to the Water Framework Directive (2000/60/EC).

Rivers discharging to the harbour are predominantly groundwater fed and this groundwater receives nitrate leached from the catchment land surface. Therefore, the geology has a fundamental influence on the hydrology of the catchment and in turn the pathways of nitrogen input to Poole Harbour. The majority of the catchment is underlain by chalk up to 300m thick (**Figure 4.6.**), with the top 50-100m being more effective in transmitting water. Outcropping chalk in this area forms rolling downland and is highly permeable. In the upper catchment and crossing the watershed, the chalk has been eroded, exposing Upper Greensand and Gault Clay. The Greensand provides spring flow to the upper Frome, while the clay provides a greater element of surface run-off in some headwater tributaries.

In the lower part of the catchment the chalk becomes confined below low permeability London Clay. The London Clay, is, in turn overlain by up to 100m of sands and clays of tertiary origin (Barton, Bracklesham and Bagshot Beds). These have a mixed permeability giving a much higher degree of surface run-off, but also localised infiltration to shallow aquifers, which discharge to small streams and the lower tributaries of the main rivers. The south part of the catchment (Corfe

River) is different in character, with the main reaches draining from a limestone plateau (Purbeck Beds) and across low permeability Wealden clay before cutting though a high chalk ridge to the Tertiary geology. Near Poole Harbour the chalk lies 100-200m below ground level.

Rainfall over the limestones, Upper Greensand, chalk and gravel geologies is readily able to infiltrate into the ground. The water moves slowly though the unsaturated formations until it enters the water table and is then transported more rapidly to outflow points, predominantly springs and into rivers as baseflow, and to the coast. This transmission process can take tens of years from the point of recharge to the outflow points. Nitrogen compounds absorbed by rainfall from the atmosphere, and then, in drainage on the land surface and through soils, also take a similar time period to be transmitted.

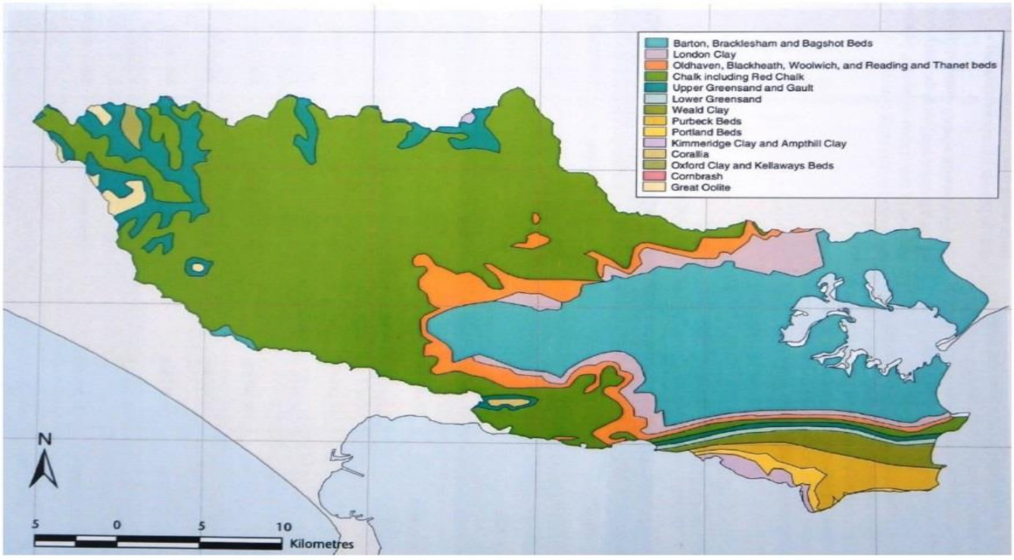


Figure 4.6.: Simplified geological map of the Poole Harbour catchment. Abstracted from the EA (2013).

Soils in the area are mainly free draining sandy and loamy soils which support a wide range of cropping and land use, including arable farms with cereals and dairying, beef and sheep production based on permanent and short term grassland and forage maize. Excluding the harbour itself, 80% of the catchment is agricultural (47% arable and 34% grassland). The remainder, mostly on the Tertiary geology is urban (10%), heath (6%) or woodland and forestry (3%). These free draining soils, especially the thinner (sandy) or shallow areas, are very prone to nitrogen (N) leaching which occurs when there is an excess of available nitrogen in the soil.

The main arable crop is wheat (c9000ha), while much smaller areas are used for maize, spring barley and oilseed rape (each c1500-2000ha). Dairy farming is a major part of the agricultural sector (c12000 head), more so than beef cattle (c4000 head). The catchment is also important for sheep rearing (c6000 head) (Defra 2005).

4.2.2. Nitrogen Pollution in the catchment- Site setting

Since 2012, Wessex Water has committed to the Poole Harbour Catchment Initiative (PHCI) - formerly known as Frome & Piddle Catchment Initiative- which involved the development of a stakeholder engagement process to identify key issues and solutions in the area.

The pilot engagement process of WSX involved investigation of the current environmental status of the watershed and identification of five key issues and pressures within its boundaries (**Table 4.1.**), with special focus on those involving non-compliance with statutory standards. According to the findings, nitrogen pollution remains the critical challenge for the Poole Harbour Catchment; thus, tackling nitrogen pollution has been identified as a priority.

Table 4.1.: Agreed key issues in the Poole Harbour Catchment (adapted from the PHCI Catchment Plan report, 2014).

Agreed key issues	Reported cause	Type
1 Nitrogen	Sewage (treatment works, CSOs, septic tanks) Agriculture (Land use management)	Current
2. Phosphorus	Sewage (treatment works, CSOs, septic tanks) Agriculture (Land use management)	
3. Sediment	Agriculture (Land use management) Highways (also acting as 'pathway')	
4. Water quantity High/Low Flow	Abstraction (water companies and agriculture) Agriculture (Land use management)	
5. Channel & Habitat alterations	Flood defence Water level management Land drainage	Historical

The Nitrogen Reduction Strategy (NRS) report for the Poole Harbour Catchment was published (in 2012 and 2013) by the Environment Agency and Natural England, who agreed to work jointly to achieve statutory and environmental objectives. NRS report provided an overview of the nitrogen sources in the catchment and the causes of failure to achieve a favourable status to date, along with a range of options for addressing the problem and an appraisal of the cost of delivering the suggested measures. Its overall aim is to ensure that discharges from future planned development within the consented levels will not have a likely significant effect on Poole Harbour.

The impact of nitrogen to the Poole Harbour is a long standing issue. A primary symptom is the growth of green seaweeds forming macroalgal mats on mudflats and among saltmarsh in intertidal areas. The extent of macroalgal mats in Poole Harbour has increased since 1980s, from a minimum of nearly 100ha to around 400ha in recent years. Mats covering over 75% of the substrate have increased from 41ha to over 200ha, a change from about 3% to around 15% of the intertidal mudflat area. The mudflat area was recorded to support macroalgal mats, with a biomass of 22 Kg/m^2 or more (EA, 2013). The macroalgal mats and their supporting element of dissolved inorganic nitrogen have resulted in the Harbour failing to meet Good Ecological Potential (WFD requirement).

The availability of plant nutrients, especially dissolved available inorganic nitrogen (DAIN), is thought to be the primary cause driving excess growth of macroalgae in estuarine environments. In Poole Harbour inorganic nitrogen shows a gradient decline from the outflow of the two main rivers of the catchment (Frome and Piddle) to the harbour entrance. Near the outflows of the rivers nitrogen levels are many times higher than the background levels in the English Channel and at the entrance of the harbour are about 5 times higher.

Inorganic nitrogen comes from many sources, conveyed to the harbour by groundwater, rivers and deposition from the air or discharges direct into the harbour. According to NRS report and information provided by WSX catchment experts during the doctoral project, in the delineated area of the Poole Harbour catchment, there are 800 farm holdings, several fish farms, 21 sewage treatment works serving a population of about 200,000, over 3500 unsewered addresses and innumerable combustion sources.

The strong rural character of the Poole Harbour Catchment is reflected in the calculated input of inorganic nitrogen, 90% of which is coming from agricultural sources and 10% from development and transport sources. A delay of 30-35 years on average between nitrate leaving the soil zone and entering the harbour is observed. This is a result of the slow travel time through the chalk aquifer. As a rule of thumb, nitrate will move 1m/yr downwards through the unsaturated zone in the Poole Harbour catchment. Nitrate will then move more rapidly (months) once it reaches the water table before entering the harbour. The mean annual input of the inorganic nitrogen to the harbour from the constituent parts of its catchment in the period 2006-2010 was estimated around 2600 tonnes per annum.

Nevertheless, a large part of this nitrogen does not reach the Poole Harbour. It is removed by biological uptake, denitrification processes in the environment and wastewater treatment. Source apportionment based on export coefficients indicates that agriculture amounts to about 84% of the nitrogen load received by the harbour (excluding the sea input). **Figure 4.5.** provides details about the estimated percentage contribution of the diverse nitrogen sources to the annual inorganic nitrogen load in Poole Harbour. As presented in the pie chart (**Figure 4.5.**) fertiliser from tillage land was identified as the single largest source at about 51%, followed by manure from livestock at about 25%. The load from development sources was 16%, with most of this coming from sewage treatment works (STWs, nearly 15%).

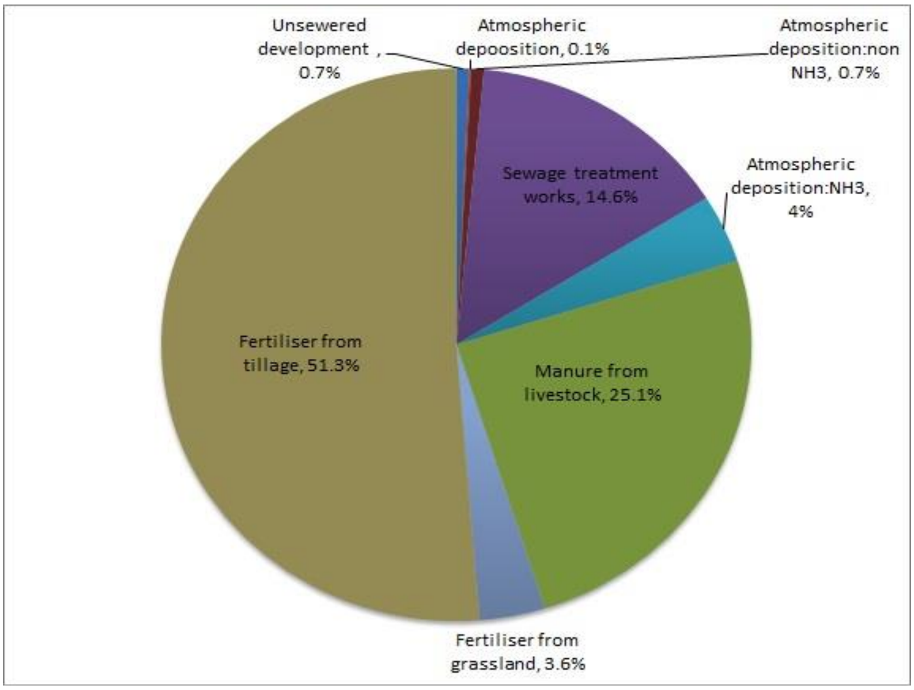


Figure 4.7.: Estimated percentage contribution of diverse nitrogen sources to the annual inorganic nitrogen load of the Poole Harbour in the period 2006-2010/11 (adapted from the EA 2013).

The two main river systems of the catchment, Frome and Piddle, are groundwater-fed (85% and 89% groundwater fed respectively) and contribute about 73% of the inorganic nitrogen load to the harbour (EA, 2013). Long-term monitoring from the Environment Agency (EA, 2013) of the water chemistry of the local river systems shows a strongly upward trend in the concentration of nitrate during the second half of the 20th century. These historic uptrends in river nitrate concentrations have been linked to macro-scale changes in agriculture and land management. As noted by the EA (2013), there is currently insufficient understanding of the relationship between the observed extent of dense macroalgal mats and data on inorganic nitrogen to confidently determine what limits to inorganic nitrogen concentrations in Poole Harbour would be needed to reduce algal mats to an acceptable level. Estimates of the nitrogen load entering the Poole Harbour reveal the increase in pollution load in the past 50 years. The computations and comparison of nitrogen loads in past and current years were used as an alternative and interim approach for informing ambition in the delivery of measures to limit the nitrogen load (Nitrogen Reduction Strategy (NRS) report, EA 2013).

The mean annual inorganic nitrogen load received by the Poole Harbour in the 5 year period 1980-1984 was selected as a basis, which was estimated to have been around 1700 tonnes. This compares with an annual load estimated at no more than about 1000 tonnes prior to 1960s. By the period 2006-2010, the mean annual load had risen to about 2300 tonnes, falling back to about 2100 tonnes with nitrogen reduction at Poole sewage treatment work (STW). Comparing the values of 1960s and 2010s, the nitrogen load has almost doubled, while a 30% raise is observed since 1980s. Nitrogen loads to the harbour are forecast (EA 2013) to continue to rise over the 15-20 years, before peaking and stabilising at around 2300 tonnes N/yr. This load reflects current input levels and constitutes a net overall forecast increase of around 215 tonnes N/yr from 2006/9.

Responding to the observed values of nitrogen load several measures have been taken to tackle the pollution in the Poole Harbour Catchment under European Directives. To date, pilot initiatives have achieved larger declines in the discharge of inorganic nitrogen from development point sources, particularly the emission of nitrogen oxides and the discharge of nitrogen from large sewerage treatment works (STWs). The largest single reduction in the harbour has come from the nitrogen removal at Poole STW, initially reducing the load to about 240 tonnes (a reduction of about 10%), while the plant was operating at 7 mg N/l discharge quality. It is estimated that, development growth in the catchment will progressively erode part of this reduction. By 2025, predicted development is calculated to add 29-41 tonnes of inorganic nitrogen per annum from STWs, and by 2035 the additional load rises to 40-71 tonnes (EA 2013).

Aiming to tackle diffuse nitrogen pollution from agriculture, several actions were initiated from diverse actors (government, commercial sectors and environmental agencies). The most successful initiative was a catchment management approach undertaken by Wessex Water on farms around boreholes for potable water abstraction, which has shown that greater reductions in nitrate leaching can be achieved at a field scale (Wessex Water 2014; EA 2013). Despite the actions taken to date in the Poole Harbour catchment area, a major decline in the river nitrate concentrations has not yet been achieved (Gooday et al. 2015; EA 2013). This is due to the combined influence of background factors, such as the long solute travel times through the chalk geology, and of the current agricultural practices affecting nitrogen management across the catchment (EA 2013).

A number of studies (Gooday et al. 2015; Ody and Martineau 2015) were recently undertaken for the Poole Harbour Catchment, aiming at shaping mitigation measures to tackle the excess of nitrogen in the watercourses, mainly caused by diffuse pollution. In all studies, it is highlighted that the travel times of groundwater should be considered in the development of the strategies addressing eutrophication in the Poole Harbour.

4.2.3. Stakeholder Position- Recommendations for the Diffuse Pollution Control in the Poole Harbour Catchment

The Nitrogen Reduction Strategy (Natural England & Environment Agency 2013) report, sets the targets for the reduction of nitrogen levels in order to prevent further deterioration in ecological status of the catchment and reduce the expansion of macroalgal growth at an acceptable level. Based on the estimates, it is recommended that nitrogen levels across the Poole Harbour Catchment should be reduced to the levels observed in 1980s (c1700 tonnes N/yr).

In order to achieve favourable conservation status under statutory requirements, it is indicated that diffuse nitrogen load will need to be reduced by an estimated 550 tonnes N/yr (Natural England & Environment Agency 2013). The target loading for the harbour could be achieved by land owners ensuring their land use activities do not exceed a maximum farm leaching of 18.3 Kg N/ha across all rural land uses. This target leaching standard provides a benchmark which will need to be applied across the whole catchment and the actual reduction will depend on current land use activities.

For point sources, an additional reduction of around 21-40 tonnes-N/yr from point sources is recommended (EA 2013). The suggestions and estimates are based the residential population forecast, in both sewered and unsewered locations of the catchment, of approximately 21,000 people by 2035 and on the on current permit conditions and sewage treatment works performance (discharge quality). The additional reduction of from point sources will be needed to ensure that the forecasted population and residential growth do not lead to a further decline in water quality. It is also determined that a limit of 10 mg/l total nitrogen (annual average) in the final effluent from Poole STW should apply, whilst five other significant STW discharges, (Dorchester, Wareham, Lytchett Minster, Blackheath, and Wool STW should be maintained at standstill provision (EA 2013).

Based on the recommended options (Gooday et al. 2015; EA 2013), improvement solutions regarding the sewage treatment options (point-source pollution) have been defined by Wessex Water, whilst land management options (diffuse pollution) were shaped from the Environment Agency. A cost-benefit analysis (CBA) was performed for both option categories. For the land management solutions, implementation measures were also listed. The costs were separated into capital and annual. Data for point source options have been provided by Wessex Water and have been converted to present value costs assuming an asset life of 20 years. For the land management measures, cost calculations were based on change in gross margin, as this could be directly linked to land areas.

Benefits from the reduction of nitrogen concentrations from both land management measures and sewage treatment works were divided into: (i) benefits for groundwater, rivers, streams and springs, (ii) benefits to Poole Harbour and (iii) benefits for ecosystems services. It is suggested (Ody and Martineau 2015; EA 2013) that the benefits will be greater for the catchment as a system where measures are taken further upstream, since longer lengths of watercourse and a greater volume of water will benefit from the reduction in nitrogen levels. The monetisation of the aforementioned benefit categories is based mainly on literature, databases and the Willingness-to Pay (WTP) approach. However, the benefits for the surface water bodies are not monetised, due to uncertainties and risk for double-counting. The low estimate of benefits provided, give an indication of which options may be economically worthwhile. The comparison of costs and benefits was associated with CO_2 reduction, biomass uptake and improvements in the quality of groundwater, surface waters and in the harbour.

A short list of the favourable solutions concludes the report (**Table 4.2.**). For point-source pollution, Wessex Water would invest in sewage treatment works improvements, aiming at a discharge of 7mg/l. In terms of mitigating diffuse nitrogen pollution, a combination of arable and livestock measures are required to ensure that the target for diffuse nitrogen reduction is achieved. If the costs of measures are divided across all farms, the most cost-effective measures would be the establishment of cover crops (winter wheat production) and the implementation of site-specific management. Concerns on the affordability of costs for the farmers are drawn. It is highlighted that a mechanism needs to be found to reduce the costs for the multiple stakeholders and eliminate the risk for future capital solutions.

More studies and reports followed the recommendations of the NRS report. The Poole Harbour Diffuse Pollution Reduction Plan was drafted in 2013 as a joint work prepared by farmers and their representatives (Wessex Water, the Environment Agency and Natural England). The plan aimed to highlight the actions that the people working and living within the boundaries of the Poole Harbour catchment will undertake in order to improve farm nutrient efficiency, reduce diffuse pollution and improve the environment. It essentially provides the detail of how the objectives of the NRS report (aiming at around 30% reduction of nitrogen losses from agriculture) will be delivered across the catchment and identifies how this work will be communicated, prioritised and incentivised.

The key measures, as recommended by the Nitrogen Reduction Strategy plan (EA 2013) and further analysed by consultancy reports (Gooday et al. 2015; Ody and Martineau 2015) were aimed at reducing leaching and improving N efficiency. For achieving these targets, actions that would limit the amount of 'available N' (nitrate NO_3-N and ammonium NH_4-N) in the soil were discussed. Therefore, farmers were advised to not apply fertilisers when plants are not growing in the autumn and winter and supply small amounts of N as crops start to grow in spring and roots are more actively taking up nutrients.

Table 4.2.: Summary of recommended strategies against the projected targets based on the outcomes of the Nitrogen Reduction Strategy report, undertaken by the Environment Agency and Natural England (2013). The quality of discharge effluent is estimated to be stabilised at 7 mg/l.

Management Strategies	Benchmark (N load, tonnes/year)	Diffuse Pollution (N load, tonnes/year)	Point-source Pollution (N load, tonnes/year)
Current status	2280	1950	330
Winter Cover Crops & controlled discharge effluent	1730	1400	330
Site-specific Management & controlled discharge effluent	1730	1400	330

For the actions recommended, the best options for mitigation of nitrate loss in the catchment were chosen after taking account of estimated costs, practicability and applicability of for farms, soil types and crop/livestock production systems representative of the area. The options were assessed using the FARMSCOPPER (FARM Scale Optimisation of Pollutant Emission Reductions) tool (Gooday et al. 2014; Gooday and Anthony 2010) which adopts the farmers' viewpoint in the estimation of costs. The likely implementation costs or potential savings, and the likely range (%) of reduction in nitrate leaching for the best options are outlined below (**Table 4.3.**).

A study correlating nitrogen leaching risk with groundwater vulnerability was performed as part of the Plan (EA 2013). It was shown that arable and pastoral farming present the highest potential risk of nutrient loss due to tillage practices, resulting in mineralisation of nitrogen, and to the limited time over which crops are growing and so, taking up nutrients. Arguably however, arable farms have the greatest potential for improving nutrient efficiencies. The latest study regarding the mitigation of nitrogen pollution in the Poole Harbour Catchment was undertaken in response to the WSX commitments under the AMP6 programme (Wessex Water Services 2014). The Scoping Study Report for Nitrogen Reduction (Ody and Martineau 2015) was performed by a consultancy firm in 2015 and explored the development of an approach for payment for ecosystem services (PES) in the catchment. The overall aim of the strategy is a load reduction of 40 tonnes of nitrogen per year.

The scoping study focussed on the level of engagement with farmers and the land management solutions required to achieve the targeted reduction. It also provided recommendations on the implementation of two different PES schemes. Approaches on payment on reduction (£/Kg N reduced) and payment by measure approach (payment based on the introduction of which of which N reduction will result) are explored and compared.

Table 4.3.: Potential savings and likely range (%) of nitrate leaching reduction for management options, as recommended by the Poole Harbour Diffuse Pollution Reduction Plan (2014) and estimated using the FARMSCOPPER tool.

Measure	% reduction N leaching	Cost £/ha	Benefits: £ saving or other
Cover crops/under-sowing: established by mid-September	3-20%	£20 - £75	↓ fertiliser N
Use fertiliser recommendation system	Up to 5%	-	£7-12/ha
Integrate fertiliser and manure nutrient supply	5-10%	-	£20-85/ha
Reduced/minimum tillage cultivations	Up to 20%	-	£10-25/ha
Increased slurry storage capacity to allow timely applications	Up to 10%	£25 – £35	↓ fertiliser N
Avoid poultry manure and slurry application at high risk times	Up to 20%	£1	↓↓ fertiliser N

Regarding the land management solutions, several measures were assessed and the option appraisal was to define the most cost effective and easy to implement approach to achieve N reduction targets. The assessment was based on the following criteria: N reduction (effectiveness), cost, co-benefits and risks, potential of farmer uptake and accountability and verifiability of the measure.

After reviewing a long list of mitigation measures for diffuse water pollution as published by Defra (2011) and gathering evidence of their local relevance from discussions with WSX and external stakeholders, a short list of six mitigation measures were selected for more detailed review (**Table 4.4.**).

The measures recommended after the assessment was performed include the reduction of applied nitrogen by 5% and the adoption of cover crops. The two scenarios were further assessed for their effectiveness to address the diffuse pollution in the catchment based on a number of assumptions (average farm size equal to 200ha, 25 farms engaged in the programme) while secondary data form literature were used.

The detailed assessment of the recommended strategies favours the 5% reduction in the applied fertiliser, as a means to achieve immediate results within the restricted timeframe and as a more transparent regime for the calculations of the N reduction. The optimum scale of implementation is the whole catchment, as the larger target area would encourage farmer engagement with the scheme. On the downside, the load reduction needs to be accounted on the entire catchment and may not be possible to be measures on a tributary (sub-catchment) scale. The travel times of the water flows are highlighted as critical for the observation of the outcomes of the implemented strategy.

Table 4.4.: Summary of mitigation measures assessed as part of the Scoping Study Report for Nitrogen Reduction against their success criteria.

Measure	N Reduction	Cost (£/t reduced)	Verification of N reduction	Uptake barriers	Co-benefits	Risks
Cover crop	Moderate	Moderate	Moderate	Low-Moderate	Decrease in P and sediment	Increased pesticide use Conflict with biodiversity targets for over wintered stubbles
Transfer of organic manure	Low	Moderate-High	Difficult	Moderate	Increased soil microbes, structure, and soil carbon	Pollution transfer
Arable reversion of woodland	High	High	Easy	High	Decreased P, sediment, GHG Increased Biodiversity, water storage, soil carbon	Long term land use change
5% reduction of applied nitrogen	Moderate	Moderate	Moderate	Low-Moderate	Decreased GHG Increased soil carbon	Seasonal variation or other management factors could cause reduced yield
Arable reversion to grassland	High	High	Moderate	High	Increased soil carbon, Decreased P, sediment, some GHG	Increased FIO, BOD, ammonia & methane
Use clover in place of N fertiliser	Moderate	Low	Difficult	Low-Moderate	Decreased GHG Increased Biodiversity	Increased release of N if grassland removed

In summary, a number of extended studies were performed since 2012, aiming to develop understanding of the catchment system of the Poole Harbour catchment and of the conditions under which augmented nitrogen loads enter its watercourses. Recommendations for mitigation of nitrogen pollution across the catchment mainly focussed on the reduction of nitrates from diffuse sources (agricultural activities). Several assessments were performed and a list of measures were assessed for their effectiveness to address the issue. The overall target of 30% reduction of total nitrogen load was set based on the analysis of nitrogen loads in past decades. For all studies undertaken after 2013, the discharge effluent from the sewerage treatment works located in the catchment is assumed at 7 mg/l N. In all studies, the favourable options for the reduction of diffuse pollution from agricultural activities were identified as: (1) the implementation of winter cover crops and (2) the reduction of the applied nitrogen across the catchment. The latter can be achieved through the implementation of precision agriculture, as recommended by the NRS report. In all studies, the role of the travel times of water flows is highlighted as critical for the observations of the outcomes and success of the implemented strategies.

Chapter 5: Holistic Asset Management & the Catchment Metabolism modelling schema

Chapter 5 defines the notion of Holistic Asset Management and demonstrates its relevance to Integrated Catchment Management. It then introduces the Catchment Metabolism modelling schema, a structured and transdisciplinary approach for modelling catchments as asset systems and serving asset management planning purposes.

The research system boundaries are drawn around the catchment, as defined in physical geography terms. The catchment is selected as the most suitable scale to assess water sustainability and the interactions among different types of capital. For the research undertaken, the catchment is defined as a hybrid, integrated asset system where both natural elements (biosphere) and infrastructure (technosphere) are included. Following the principle of integrated water resources management and ecosystem services (Cook and Spray 2012), the ecosystem is considered as a stakeholder who plays an active role within the boundaries of the catchment. The research suggests that Holistic Asset Management (HAM) at a catchment scale is the key for effective and sustainable management of water resources. HAM introduces a novel approach to asset management in the water sector, which includes the three sustainability pillars (People, Planet, Profit) and is presented in a format that is easy to apply and communicate. It enables a systemic view on water and asset management strategies and the involvement of multiple stakeholders. Thus, HAM enables in practice to tackle of the pre-identified key issues of a catchment 'synergistically' (**Figure 5.1.**).

To translate the Holistic Asset Management rationale into a modelling approach, a number of well-established tools and methods from various disciplines are synthesised based on their suitability to serve the research goal (section 5.2.). The whole-system approach developed in this thesis is based on the principles of Integrated Catchment Management (ICM), Water Accounting (WA) and Environmental Multi-Regional Input-Output (E-MRIO) analysis. It builds on a combination of concepts and methods that have been reviewed and approved for their ability to address sustainability issues (Little et al. 2016; Ma et al. 2015; Paterson et al. 2015; Xu et al. 2015; Rudell et al. 2014), and shape optimised planning strategies (Ma et al. 2015; Rudell et al. 2014; Daniels et al. 2011) for better resource efficiency. The Catchment Metabolism (CM) schema offers an approach where researchers and end users can conceptualise catchment systems and their processes, which is essential for integrated water resources management (Macleod et al. 2007).

The constructed synergies form the platform for integrating natural capital in the strategic planning schemes of the water industry.

This chapter describes the main outputs of the research undertaken. The CM modelling schema responds to the need for evidenced based approaches, which can be used in the practical application of sustainability and systems thinking principles in the water industry. It is tailored to address current challenges of the water sector and its design enables practitioners to apply research advancements. One of the advantages of the schema is that systems-thinking is required, hence, collaboration among experts within the water sector occurs. This reflects the transdisciplinary nature of the work.

The chapter is organised as follows: after the introductory note which explains the rationale of the research and re-affirms its system boundaries, the creative process and rationale for identifying the appropriate techniques used to formulate the underpinning methodology are described. The synthesised approach is then presented and the CM modelling schema is illustrated through its application to the selected example catchment, as described in Chapter 4. The chapter concludes by discussing the steps for the practical adoption of the schema in the UK water industry.

5.1. The underpinning rationale of the Catchment Metabolism modelling schema

The section gives an overview of the rationale of the creation of the modelling schema and its underpinning concepts and tools. The explanatory brainstorming diagram outlines the synthesis of the transdisciplinary methodology (**Figure 5.2**). The divergence of the work and the lack of previous relevant approaches in the field of asset management required a comprehensive literature review to be performed. This mainly focussed on identifying and analysing the tools for integrated environmental-economic accounting widely used in other fields and been applied in different scales (e.g. infrastructure asset systems, community, city).

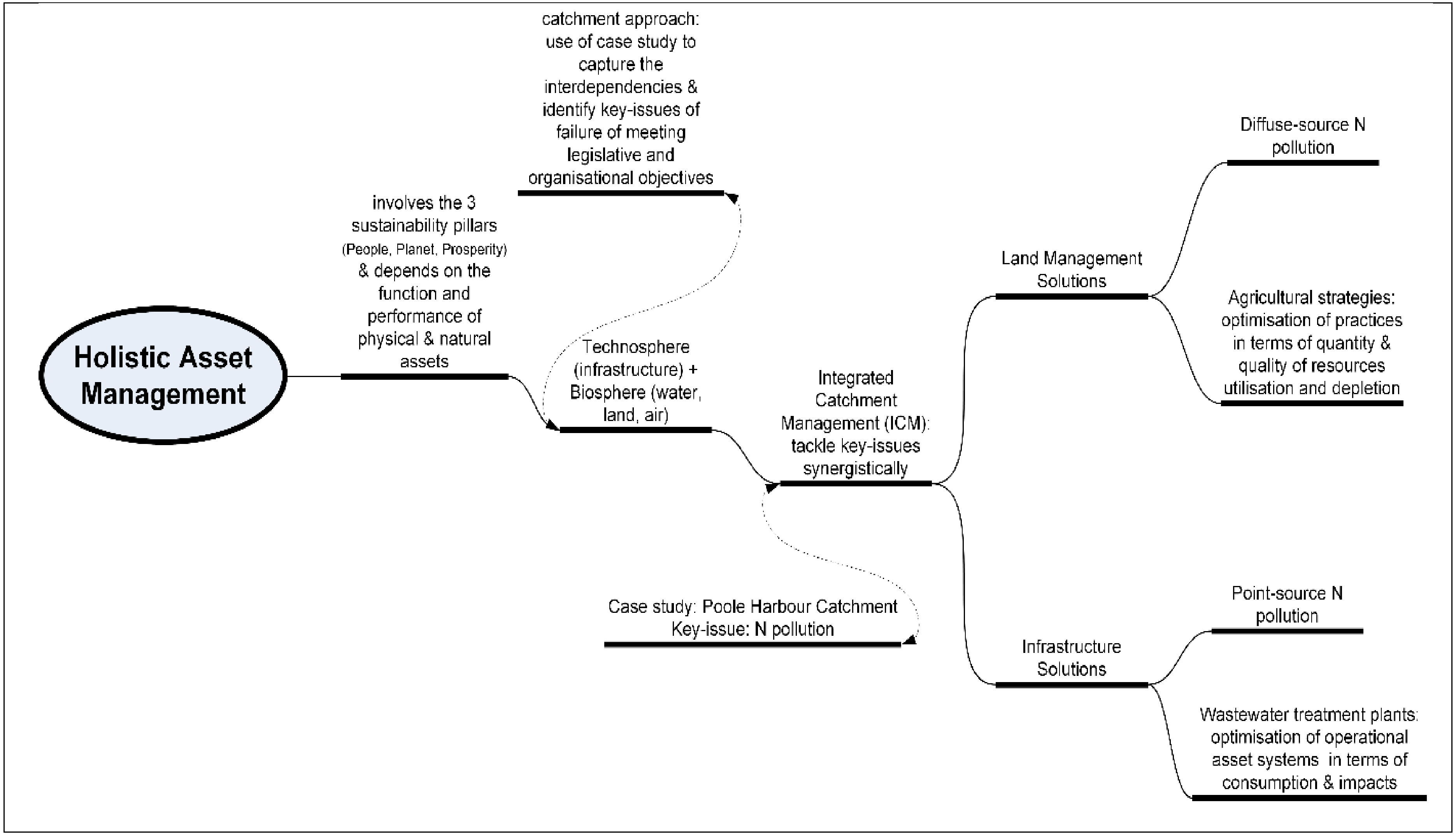


Figure 5.1.: The rationale of the research: holistic asset management through integrated catchment management.

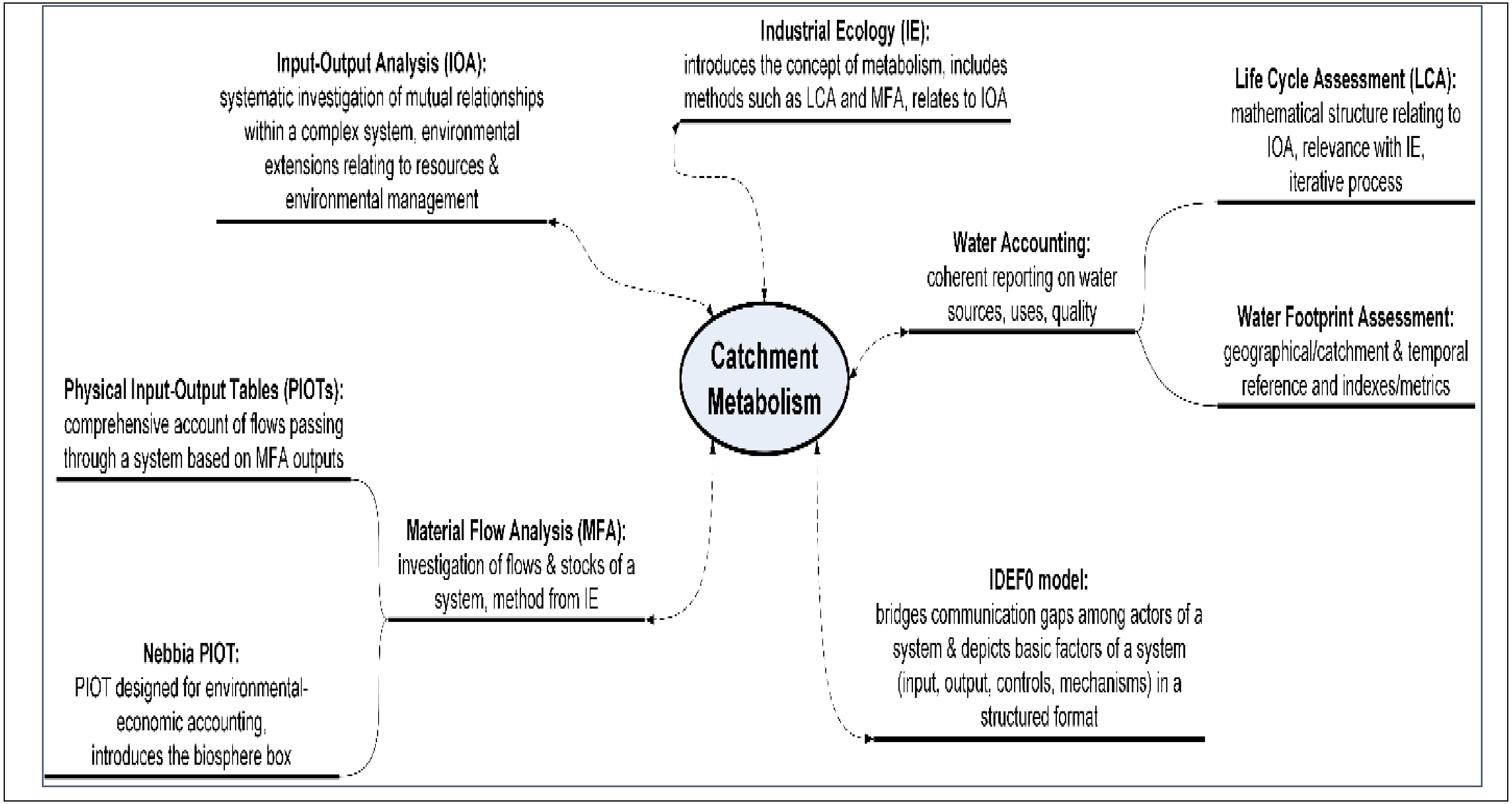


Figure 5.2.: The formulation of the Catchment Metabolism modelling schema based on a robust synthesis of methods available from systems engineering and environmental-economic accounting.

For the formulation of the CM, it was hypothesised that the currently analysed tools could be applied for the creation of catchment-based approaches for asset management purposes. For the hypothesis to be held true the tools need to account for both the natural and the built capital on a catchment basis.

The original intention was to create an approach using life cycle management and Life Cycle Assessment at a catchment scale to achieve the research goal, i.e. the creation of a catchment-based modelling schema for the realisation of holistic asset management from the water industry. To overcome the limitations of LCA in terms of its spatial reference and applicability at delineated geographical areas (Baumann and Tillman 2004), a number of other tools were explored. Industrial Ecology (IE) -which is the research field underpinning LCA- was examined to determine how it could be used for the creation of the CM schema. In order to do this, the development of the field of IE into other widely used concepts was explored using a detailed literature search. Four main techniques were identified: Water Accounting, Input-Output Analysis (IOA), Material Flows Analysis (MFA) and IDEF0. The structures and main knowledge blocks of a number of concepts and tools were analysed and then synthesised based on their strengths and contributions to specific objectives of the modelling schema. The overview of the concepts and techniques is presented in the following section (**section 5.2.**), along with the linkages among them.

The concept of metabolism derives from the field of Industrial Ecology and has been used as the conceptual basis of the modelling schema. Material Flow Analysis (MFA) and its Physical Input-Output Tables (PIOTs) formulate the reasoning for flow accounting within the catchment systems and construct the format of the Catchment PIOT. Input-Output Analysis (IOA) and its environmental extensions are used as tools to account for the multiple flows of the complex catchment system in a constructed approach. Water Accounting methods provide the metrics for water flow accounting in multiple systems. The IDEF0 model has been selected to serve as a method to collect and depict information for the subsystems of the catchment and to bridge communication gaps among the experts involved in the process of integrated catchment management.

5.2. Overview of the techniques formulating the Catchment Metabolism modelling schema

Life Cycle Thinking and its methods initiated the underpinning methodology. Despite their conceptual strengths, their application at a catchment scale is rather challenging. Therefore, a 'retrospective' approach was adopted. This included the identification of the origins of the methods widely used in Life Cycle Management, along with the thorough study and analysis of their principles, building blocks and commons applications. Building a new modelling schema on the same principles underpinning LCA or LCC would provide a robust basis that would enable the application of Life Cycle Thinking at a catchment scale. It would also shed light on the improvements necessary to apply well-known tools at new research fields, such as Integrated Catchment Management. The origins of the Life Cycle Management tools were identified in the field of Industrial Ecology (Ehrenfeld 2004; Korhonen 2004; den Hond 2000.)

5.2.1. Industrial Ecology & Metabolism

The field of Industrial Ecology (IE) states the analogy between the industrial system (anthroposphere or technosphere) and the natural environment (biosphere) and consists a framework towards practical sustainability. It has been applied for the optimisation of material cycles within the industrial systems as it serves for the development of symbiotic relationships among industries. IE offers a comprehensive, integrated view of the components of the industrial economy and their relationship with the biosphere. IE emphasises on the biophysical basis of the human activities; thus, on the complex patterns of material flows within and outside the industrial system (Ehrenfeld 2004; Korhonen 2004; Brattebø 2003; Erkman 2003; den Hond 2000). IE treats the industrial system as a complex organism with unique metabolic rules (Suh and Kagawa 2005).

The basic methodologic concept of IE is that of ‘industrial metabolism’, which is a descriptive and analytical concept based on the principle of the conservation of mass applied for the understanding of the complex patterns and dynamics of flow and stocks of material and energy within the industrial system. Industrial Metabolism has been widely applied in the urban context, as summarised by Clift et al. (2015) and involves a range of methods (e.g. Life Cycle Assessment, Material Flow Analysis) which have served planning and development purposes especially in the form of regional flow analysis (Brattebø 2003; Erkman 2003; den Hond 2000). The concept of ‘metabolism’ implies the need for a systemic perspective: it brings in the totality of a techno-economic social system within a natural environment (Fischer-Kowalski 2003)

The term ‘metabolism’ is, by nature, an interdisciplinary enterprise (Fischer-Kowalski 2003). It appeared in 1860s, both as applied to organisms and to human social systems. The modern biochemical notion refers more to the transformative processes of cells, organs and organisms (Oxford Dictionary of Biochemistry and Molecular Biology) and does not focus on the environment-organism interface. In ecology, the term ‘metabolism’ is used to refer the energy conversion and nutrient cycling in ecosystems (e.g. Humphries and McCann 2013; Brown et al. 2004). What is common to the biochemical and the ecological approach is the idea of metabolism being a complex self-organising process of autopoietic (i.e. capable of reproducing and maintaining itself) systems, dependent on the characteristics of the system (**Figure 5.3.**).

The value of Industrial Ecology and Metabolism for the management of aquatic systems and strategic sustainable development has long been recognised (Korhonen 2004; Billen 2003). Nevertheless, its applications in water-related studies is rather limited (Núñez et al. 2010). Recent water-related IE applications focus on the development of indicators for effective water management (Ziolkowska and Ziolkowski 2016; Farreny et al. 2013), the formulation of models for water demand and pricing (Dharmaratna and Harris 2012; Morales-Pinzón et al. 2012) or the environmental assessment of municipal and urban systems (Lemos et al. 2013; Oliver-Solà et al. 2013) and cultural services (Farreny et al. 2012).

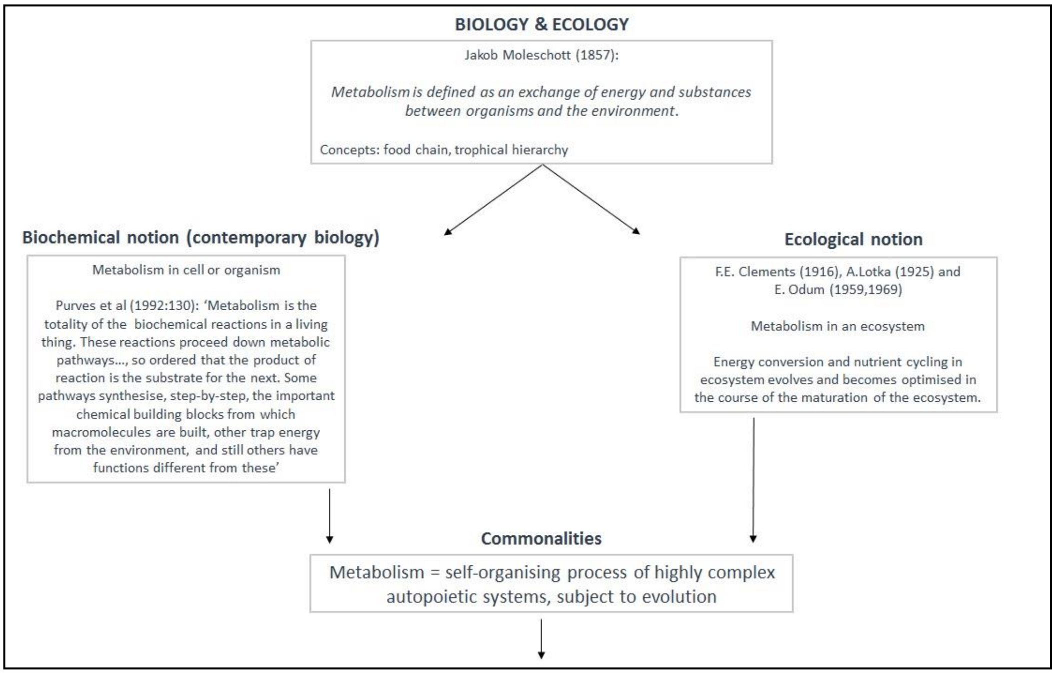


Figure 5.3.: The milestones of 'metabolism' in biology and ecology. Adapted from M. Fischer-Kowalski, 2003.

The research field of industrial ecology comprises of several methods that have been developed to analyse parts of the technosphere (van der Voet 2011). The growing sophistication of the IE research urges for more systemic empirical work which would move theory and methodology forward (Lifset 2013). Billen (2003) advocates that if the approach of industrial ecology was integrated into the scope of scientific ecology, a major step would be taken towards achieving a general science of the functioning of human-affected environmental systems. He stresses the need of knowledge synthesis and integration for the scope of regional studies, partially for those intended as a basis for the management of water systems (e.g. rivers, estuaries, coastal zones) where the effect of non-point pollution sources urges for integrated management solutions.

5.2.2. *Water Accounting & Industrial Ecology*

The field of Water Accounting (WA), as introduced earlier (Chapter 2), is only loosely related to the field of Industrial Ecology. The connections lie in the recent methodological advancements of LCA, which enables the development of detailed water accounts and the performance of water-specific environmental assessments (e.g. Kounina et al. 2013). These developments have emerged in the academic literature in the form of Water Inventories, which employ computational frameworks and indicators. The indicators largely relate to the concept of 'Water Footprint', as introduced by Hoekstra (2003) and as defined in the ISO standard 14046:2014. Further details on Water Inventories to follow at a later stage (Chapter 6).

The parallel development of the two methodologies in the field of Water Inventories, namely LCAwater and Water Footprint Assessment (WFA) has mobilised a vast amount of literature, with a number of review (Kounina et al. 2013; Berger et al. 2010) and critique (Wichelns 2015; Chenoweth et al. 2014; Tillotson et al. 2014; Yang et al. 2013) papers being published over the last few years. Attempts to pursue methodological harmonisation between LCA and footprint research are strongly encouraged in the literature.

Recent case studies (e.g. Zhi et al. 2014; Feng et al. 2011; Yang et al. 2010, Yu et al. 2010) have focussed on the combined use of water footprint with Input-Output Analysis (IOA) as a means to inform regional or national decision-making. IOA also underpins the Water Accounting international frameworks (Pedro-Monzonís et al. 2016a).

5.2.3. *Input-Output Analysis*

Wassily W. Leontief (Nobel Prize winner in economic sciences, 1973) developed the method so called Input-Output Analysis (IOA) when he searched for analytic tools to investigate the economic transactions between the various sectors of an economy. It is a method for systemically quantifying the mutual interrelationships among the various sectors of a complex economic system. It connects goods, production processes, deliveries, and demand in a stationary as well as dynamic way. The production system is described as a system of flows of goods (provisions) between the various production sectors. Since its development in 1930s, IOA has been further developed and applied to a large number of studies and fields (Feng et al. 2011) and has proven a useful tool used for planning in market and centrally planned economies. Input-output analysis is a mature scientific field, which has had the ambition to facilitate interdisciplinary research, by connecting different disciplines. From a practical perspective, input-output tables provide a valuable compilation of statistical data at a national or sectorial level which could be used in industrial ecology studies (Suh and Kagawa 2005).

The basic input-output model is explained below. The mathematical structure of an input-output system consisting of n linear equations is shown as in Equation (5.1). The equation depicts how the production of an economy depends on inter-sectoral relations and final demand.

$$
\begin{aligned}
X_1 &= z_{11} + z_{12} + \cdots + z_{1n} + Y_1 \\
X_2 &= z_{21} + z_{22} + \cdots + z_{2n} + Y_2 \\
\dots & \quad \dots \quad \dots \quad \dots \quad \dots \\
X_n &= z_{n1} + z_{n2} + \cdots + z_{nn} + Y_n
\end{aligned} \tag{5.1}
$$

where n is the number of economic sectors of an economy; x represents the total economic output of the i^{th} sector; Y represents the total final demand for the product of the i^{th} sector and z represents the interindustry sales of the i^{th} sector.

Suh and Kagawa (2005) acknowledge the communalities of intellectual grounds shared between Input-Output Analysis and Industrial Ecology. Both approaches endorse a system's view and place strong emphasis on developing sounds empirical knowledge based on real-world data. By adopting a broad perspective, they intend to tackle the limitations related to partial analyses and provide 'alternative' approaches for managerial and policy decision-making.

Economic input-output modelling has also been used for environmental systems analysis. Environmental input-output analysis (E-IO) and its multi-regional extensions (Environmental Multi-Regional Input-Output E-MRIO) have emerged as popular and promising frameworks for sustainability analysis (Wiedmann et al. 2011; Hendrickson et al. 2007). E-IO enables assessment of natural resources and pollutants embodied into goods and services and in their supply chains along the economy. The significant differences between localised environmental issues associated with water use and trans-boundary issues calls for special attention to how E-MRIO can help understand and best manage freshwater resources (Wiedmann et al. 2011). Multi-regional input-output (MRIO) analysis enhances this capability by mapping the geography of the resource use, emissions and other environmental effects and provides a spatially-explicit framework than can assist in assessing environmental impacts. This ability of 'geo-position' is vital for assessing sustainable scale and impacts for many environmental resources, especially for water, since its sustainability and management is considered at a local level (Daniels et al. 2011). Recent research (Rudell et al. 2014; Zhi et al. 2014; Feng et al. 2011; Yang et al. 2010, Yu et al. 2010) shows progress in the integration of geographical information and process-based water footprints (WFs) in input-output models and accounting tables. The joint use of E-MRIO complements water stress indices

(WSI) by providing detailed mapping of the consumption to production and trade-off flow pathways, as it utilises and links economic and environmental data from across several regions. Daniels et al (2011) suggest that further research is necessary in order to align the functional features of E-MRIO upon the spatial, catchment focus of WSIs and take advantage of their combined use.

5.2.4. Physical Input-Output Tables & Material Flow Analysis

Physical Input-Output Tables (PIOTs) are accounting tools which provide a comprehensive description of anthropogenic material flows (e.g. material and energy flows) passing through the economy of a country. For their construction, the mass balance principle is utilised and the economic system is depicted as being embedded in the larger natural system. A Material Flow Analysis (MFA) study can form the basis for the quantitative information necessary to construct a PIOT (De Marco et al. 2009). MFA has been widely applied for assessing the material base and resource throughput the national economies (Giljum and Hubacek 2009; Brunner and Rechburger 2003) and its applications mainly include the quantification of aggregated resource inputs and outputs of economic systems and are performed according to its methodological guidebook (EUROSTAT 2001). Physical Input-Output Tables are constructed based on the principles of the Monetary Input-Output Tables (MIOT), which assume a closed economic system, at a national scale. Thus, the flows traditionally depicted in a PIOT concern only the flows inside the national territory.

The result of the transferral of MFA data to the PIOT is that the output produced by each production chain is split among various columns, where each column refers to a specific economic sector. A full PIOT can show the material flows between sectors (industry by industry) or the materials required to transform other materials in the production process (materials by materials or commodity by commodity). In general, a PIOT is a tabular scheme in which a certain number of economic activities or sectors are represented by their material input and output. Nebbia (2000; 1975) outlines a type of PIOT aiming to capture the circularity of industrial metabolism in terms of a “natural history of commodities” – from the environment, and back to the environment. At the heart of Nebbia’s PIOT is an economic-ecologic accounting carried out by the principles of commodity science to determine the intersectoral flows between and within the biosphere and the technosphere. The distinguishing feature of this approach is that also the biosphere, not just an economic system is divided in sectors, between which intersectoral flows may occur. As analysed in De Marco et al. (2009), the general formation for the construction of a Nebbia’s PIOT can be synthesised in a table which is initially split in four different quadrants:

	Nature (i)	Technosphere (j)
Nature (i)	Aii	aij
Technosphere (j)	Aji	ajj

where aii represents flows within the biosphere, aji resources 'sold' from the biosphere to the technosphere (e.g. water used in production processes), aji material flows from the technosphere to the biosphere (e.g. waste disposed or emissions) and ajj commodities exchanged between different technosphere sectors (e.g. electricity 'sold' to production processes).

From this PIOT one can compute the 'physical' mass of materials absorbed by final consumption, including exports and stocks, minus the imports. However, its application to date excludes the mass of water which circulates through the natural and economic systems (e.g. embedded water in products). The major shortcoming of PIOTs is that all flows are accounted in one single unit; thus, the consideration of the qualitative differences of materials flows in terms of different environmental impacts is very limited (Giljum and Hubacek 2009) and more research needs to be undertaken to overcome this issue.

5.1.5. The IDEF0 model

Undertaking the steps to construct a PIOT that would represent outputs of the sectors within the complex catchment system, a tremendous amount of data is required, along with the contribution of multiple experts. To overcome this challenge, a functional modelling language- IDEF0- is introduced in the schema. IDEF0 (a compound acronym deriving from Icam DEFinition for Function Modelling, National Institute for Standards and Technology, 21 December 1993) is a method initially designed to model manufacturing processes. Nevertheless, its theoretical basis allows its use for modelling the decisions, actions, and activities of an organisation or a system. It has been applied, but is not limited, to topics such as strategic planning, hybrid systems design and business process reengineering (Feldmann 1998) and has proven useful for handling complexity and bridging communications gaps between various actors involved in a system. Recent research (Settanni et al. 2015, 2014; Šerifi et al. 2009) highlights the applicability of the method across disciplines and sectors, for the development of modelling approaches for product service systems (PSS), for measuring performance and outcomes of asset systems and for designing software packages.

An IDEF0 model (made of several IDEF0 diagrams) depicts constraint, not flow. The graphical elements of IDEF0 are very simple (**Figure 5.3.**); just boxes and arrows. The syntax and semantics for both IDEF0 diagrams and models are precisely defined in the Federal Information Process

Standards for IDEF0 (FIPS PUB 1983). Each activity box on an IDEF0 diagram depicts the function described by the verb phrase written in the box. The boxes represent actions, whereas the arrows shown entering and leaving the boxes represent interfaces and depict things that are needed or produced by the function. Unlike data flow diagrams, IDEF0 model shows what controls each activity and who performs it, as well as the resources needed by each activity. Developing an IDEF0 model is a step-by-step procedure which begins at the point which the author determines the basic model parameters: the purpose and the viewpoint. For the same system, different IDEF0 models can be created, based on the selected viewpoint. As such, multiple IDEF0 models can be constructed for a catchment system, with the input from both an asset and a catchment manager. For each of the cases, the perspective adopted would impact on the information included in the model. The expertise and area of interest would emphasise on some aspects of the catchment system (e.g. built assets or ecosystem for the asset and catchment managers respectively). Only the synthesis of such IDEF0 models would enable the creation of whole-systems' IDEF0 model, which would be inclusive of the information or details of all the sub-systems of a catchment.

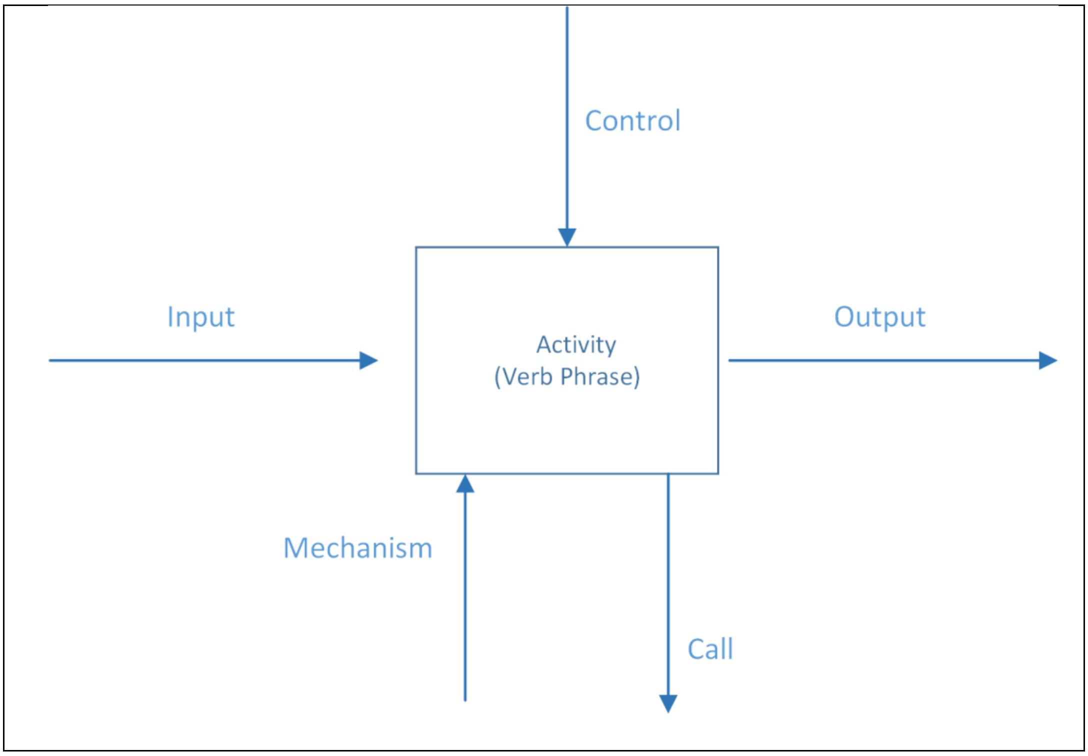


Figure 5.4.: IDEF0 Activity Box and basic syntax.

5.3. The Catchment Metabolism in practice

The Catchment Metabolism modelling schema is created using a transdisciplinary approach which synthesises a number of techniques. The schema is a structured modelling approach represents the catchment as an asset system. The process of its application is illustrated in this section, using the Poole Harbour Catchment as an example.

5.3.1. Constructing the Catchment Physical Input-Output Table: a step-by-step process

The creation of the Catchment Metabolism (CM) schema is based on the combined use of the concepts and tools as analysed in the precious section. For the needs of the research, the notion of 'metabolism' refers "inter-industrial" relationships taking place within the system's boundaries; thus, to the activities and inter-relations of the water-actors of the catchment, which affect the water cycles taking place within those spatial boundaries.

A number of steps are undertaken in order to depict and map the metabolism of the selected system. The Catchment Physical Input Output Table (C-PIOT) is constructed through a sequel of interlinked stages which add value to the modelling schema. The C-PIOT is developed as a structured way to map the metabolism a catchment. The metabolic relationships of the catchment compartments are mapped over a period of a year. This time scale has been chosen in order to serve practical and scientific purposes and also comply with the rules of the original PIOTs. The Catchment PIOTs can also be constructed for the wet and dry periods of each year, so that variations of the flows circulated in the system are depicted.

In order to gain insight in the natural processes occurring within the selected scale, the breakdown of the biosphere in its metabolic compartments is introduced in the C-PIOT, following the terminology of MFA. Therefore, the quadrant aii – which represents the flows within nature – is split into: Atmosphere (Air), Hydrosphere (Water), Pedosphere (Soil) and Lithosphere (Geology). This alteration provides a better understanding of the natural occurring processes of the ecosystem of a catchment which affect its economic activities, e.g. agriculture. As a result, one can fit in the PIOT the water volumes circulated within the catchment system; the water flows circulating in both biosphere and technosphere.

Following the example of the original PIOT, the first step to the construction of the Catchment PIOT is the performance of a Material Flow Analysis (MFA) of the catchment. A modified flow chart (**Figure 5.14.**) describes the catchment as an integrated system, based on the consequential relationships among its elements. Its focus is the water circulation within the system boundaries which assists in explaining the relations and interdependencies among its subsystems, both natural and artificial, serving mainly information display and communication purposes. Studying the water circulation allows for the identification of the main water-related activities which take place within the catchment's boundaries and their actors.

For the selected case study (Poole Harbour Catchment, Chapter 4), we observe the presence of three main water-related ‘industries’: Ecosystem, Water Company, Agriculture. In this catchment, the origin of water available for use in the technosphere (urban water cycle, agriculture) is mainly groundwater; surface water flows are also accounted because of the import of water trade volumes from adjacent catchments. These two activity categories produce different wastewater, in terms of its quality and quantity, as well as character, referring to point and diffuse pollution respectively. The quality of the return flows to the aquifers strongly depends on the intensity of agricultural activities. The infiltrated water is then abstracted to re-participate in the water cycle and its quality, mainly in terms on nutrient load influences the intensity of the water treatment process, especially in relation to the energy consumed.

After the identification of the key water actors and the role of their activities within the catchment boundaries, the metabolism of the most critical subsystems needs to be studied. The criticality of the subsystems selected reflect both the scope of the work and the key-issues in the designated catchment.

IDEF0 diagrams are produced for each the identified “industries” or actors, analysing the inputs, outputs, controls and mechanisms of their subsystems. The IDEF0 diagrams for all the actors of the catchment are produced as part of the analysis. The IDEF0 model analyses the subsystems of the catchment system and gives an overview of their main attributes: inputs, outputs, mechanisms and controls.

In the first top-level diagram (A-0) the purpose and the viewpoint of the model are stated (**Figure 5.5.**). For the research undertaken, the scope of the IDEF0 model is to describe the attributes and anatomy of a sustainable catchment system. The viewpoint adopted is that of an asset or catchment manager/expert of a water company.

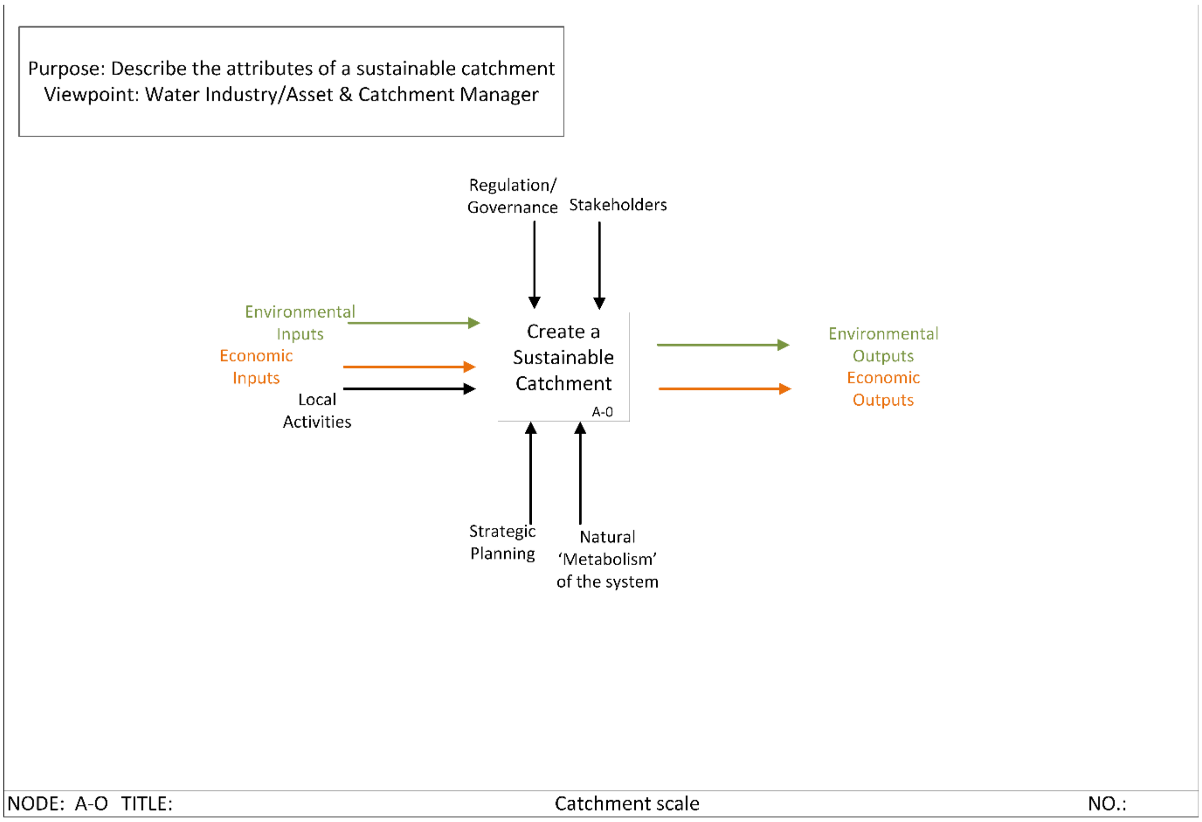


Figure 5.5.: The top-level IDEF0 diagram (A-0). Describes the overall aim of the IDEF0 model and the perspective adopted for its creation.

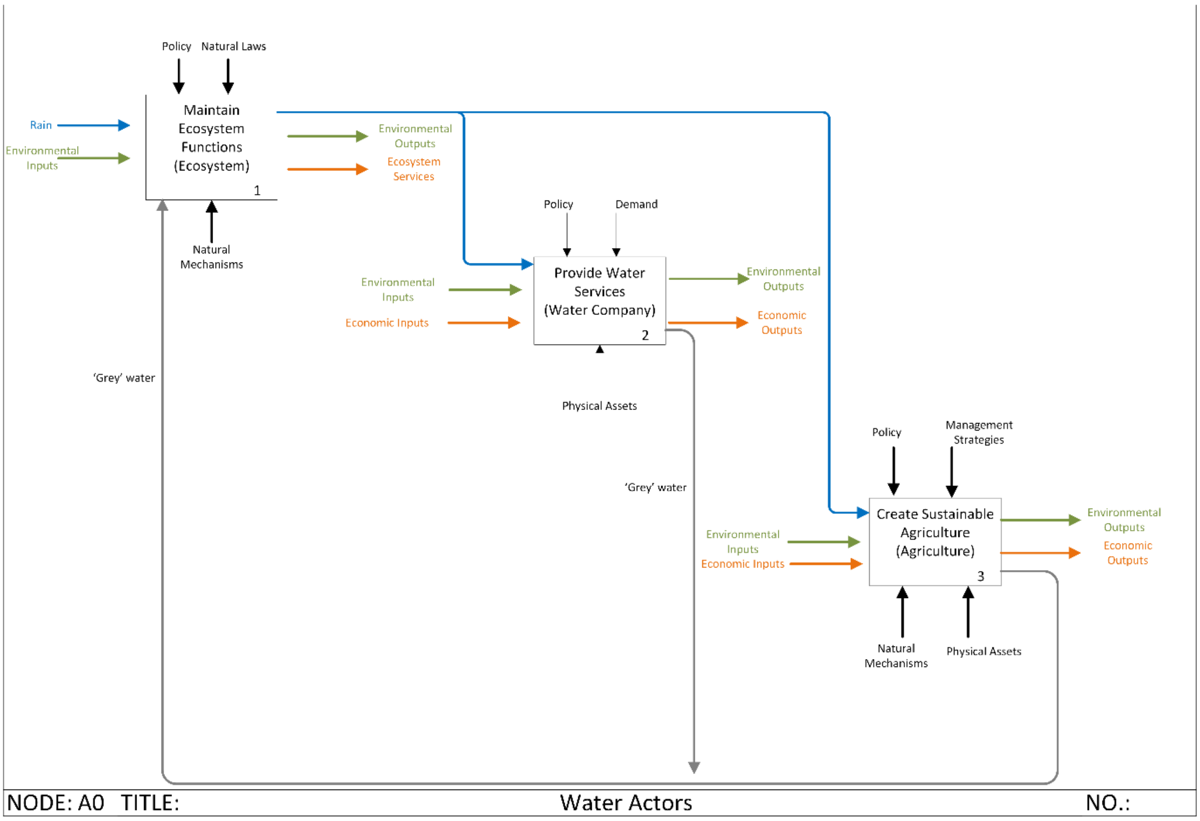


Figure 5.6.: In the A0 IDEF0 diagram the main water actors of the example catchment are depicted: the ecosystem, the water company and the agricultural sector. Each of the three constitute the core of their own subsystems, although interlinkages exist.

Then, the main actors and their contributions towards achieving the scope of the model are presented (AO) (**Figure 5.6.**). For the actor ‘Ecosystem’, maintaining the ecosystem will ensure the provision of the ecosystem services. For the actors ‘Water Company’ and ‘Agriculture’ the provision of water services and the creation of a sustainable agricultural subsystem are their contributions respectively.

In the next part of the model (A1), the focus shifts to the internal anatomy of the actors involved. For the actor ‘Ecosystem’ the three natural cycles (water, carbon, nitrogen) are depicted. The diagram does not show the complex interlinkages among the three cycles, as it is considered out of the scope of the research undertaken. Each of the cycles represent a subsystem of the actor ‘Ecosystem’, which produces environmental outputs and ecosystem services for the benefit of the biosphere and the technosphere respectively.

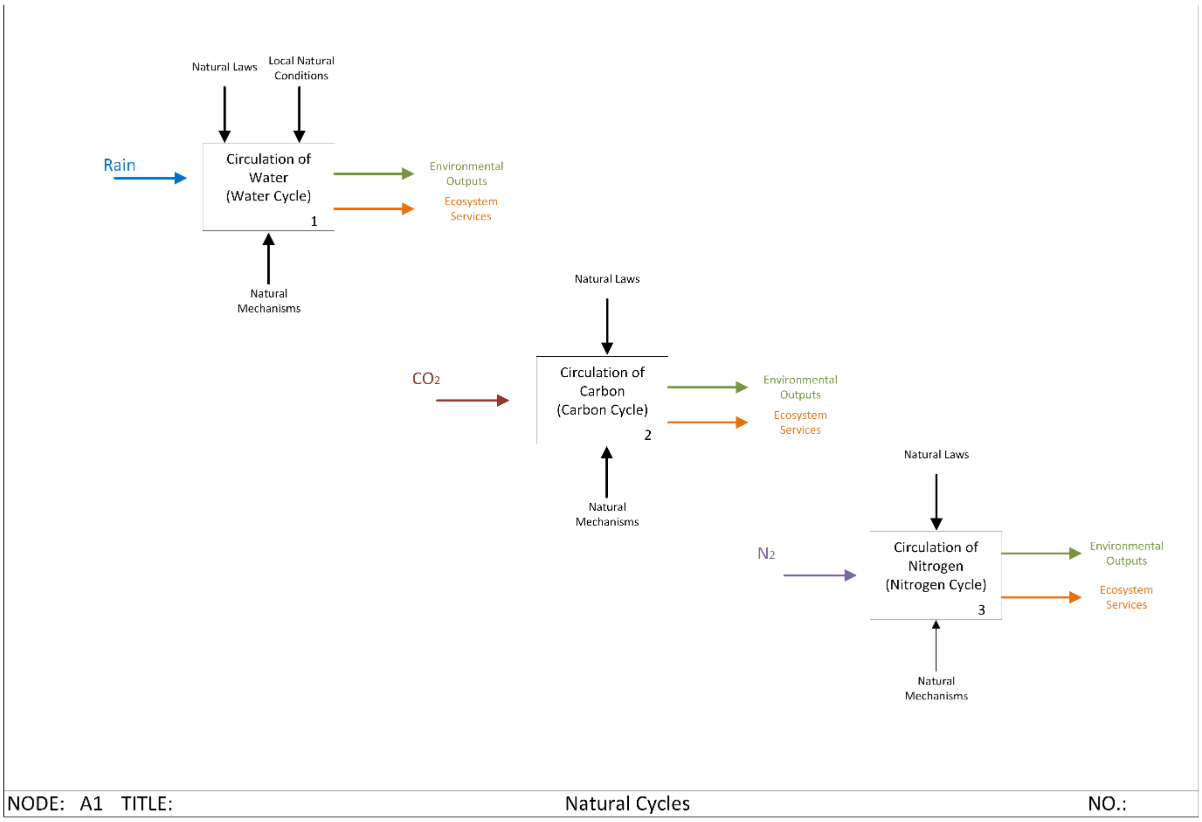


Figure 5.7.: The A1 IDEF0 diagram, depicting the analysis of the actor ‘ecosystem’ in the three natural cycles: water, carbon, nitrogen.

In the latter part of the IDEF0 for the actor “Ecosystem”, the further analysis focusses on the investigation of the water cycle as the main ecosystem function (A11). Same principles and representations would apply to the other natural cycles occurring in the catchment boundaries. The life cycles or their stages are broken down into the involved sectors, resulting in a pictorial factor analysis. For instance, for the natural water cycle as the focal point, the processes (evapotranspiration, percolation, infiltration, run-off) occurring within the subsets of the natural environment (atmosphere, pedosphere, lithosphere, hydrosphere respectively) are demonstrated followed by the factors that control the natural process (e.g. the porosity of the lithological formation controls the volume of the water infiltrated) and the mechanisms that result in the natural output (e.g. the capillary mechanism drives percolation). This latter part of the IDEF0 model (A11) shapes the Catchment PIOT, as the sectors and their processes formulate the columns of the produced table. Also, the information/data from the IDEF0 model are transferred in the tabular format to build a sector by sector (sector x sector) PIOT (**Table 5.1.**).

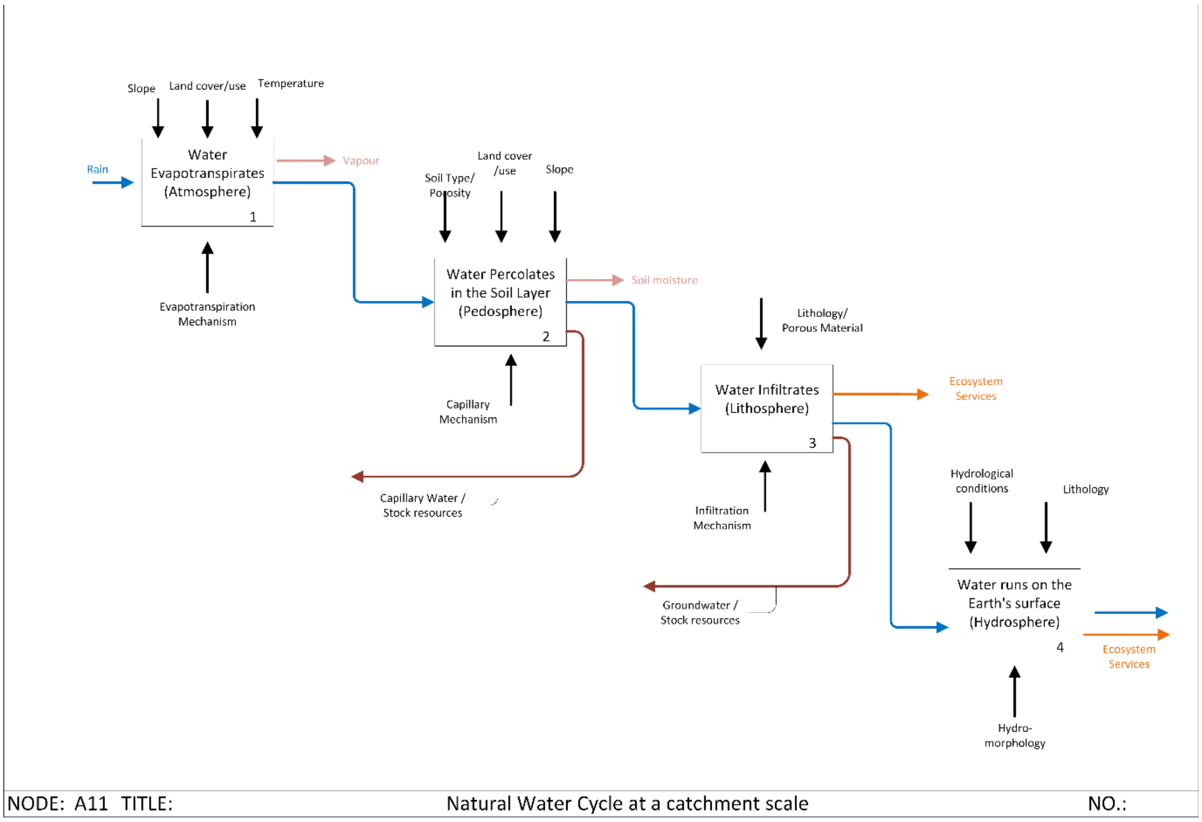


Figure 5.8.: The A11 IDEF0 diagram, analysing the natural water cycle as occurring at a catchment scale. The anatomy of the ecosystem involves its main sectors: atmosphere, pedosphere, lithosphere and hydrosphere. The outputs of this diagram form the ground of the Catchment PIOT for the actor 'ecosystem'.

The IDEF0 model continues with the analysis of the actor 'Water Company' (A2). The main activity (Water Services) is assumed to be identical with the Urban Water Cycle. As such, the sectors identified for this actor are the stages of the urban water cycle (abstraction, water treatment, distribution, wastewater treatment). The environmental and economic inputs for these sectors are normally externalities to the catchment system; for example, the chemical used for the treatment processes are produced elsewhere and then imported in the catchment. The main mechanisms identified in this subsystem are the physical assets; thus, the infrastructure owned and operated by the water company to deliver its services. The water demand and the multiple health and environmental regulations control the environmental and economic outputs of the stages of the urban water cycle. For this actor, no further analysis needs to be performed. The A2 diagram forms the basis for the Catchment PIOT water services' section.

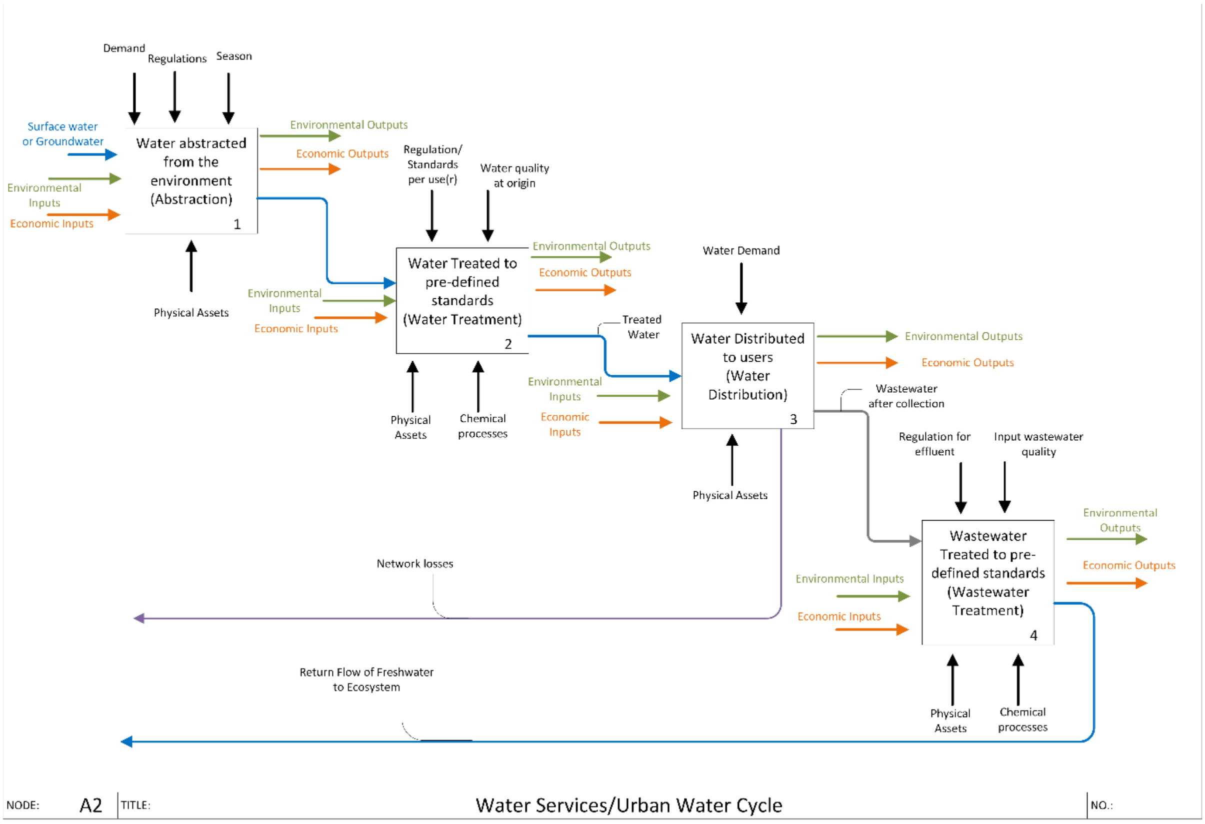


Figure 5.9.: The A2 IDEF0 diagram analysing the anatomy of the actor 'water company'. The sectors of this actor are identical with the stages of the urban water cycle. This diagram formulates the ground for the Catchment PIOT of the actor 'Water Company'.

The actor 'Agriculture' is then analysed (A3). Two main activities are identified for the selected catchment: annual cropping and farming (referring to livestock). The hybrid anatomy of the agricultural sector – in terms of the contribution of both biosphere and technosphere elements in the delivery of services- results in the increased complexity of the diagrammatic representation. The activities appear interlinked. The interconnections are dependent on individual practices or implemented policies. For both activities, the environmental and economic inputs can be considered as externalities to the catchment system. The same applies to the outputs, as they are traded to other regions. The hybrid anatomy of the actor 'Agriculture' is also evident in the combined influence of natural mechanisms and physical assets in the delivery of the services. The regional natural conditions, policy requirements and incentives and the market opportunities are identified as the main control factors of the outputs of this subsystem.

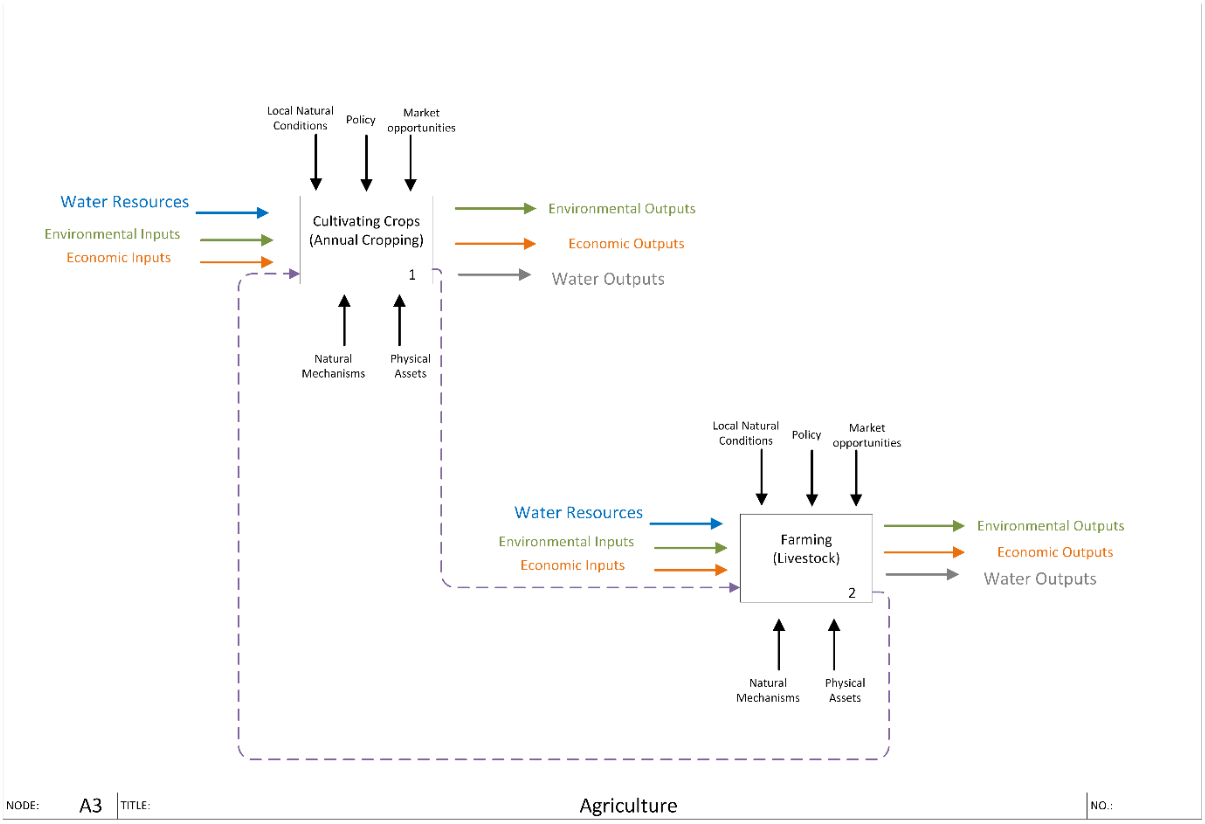


Figure 5.10.: The A3 IDEF0 diagram depicting the two main activities of the actor 'Agriculture'. Each of the activities create a separate subsystem; interconnections depend on individual practices or implemented policies.

The further analysis of the analysis of the actor 'Agriculture' involves the investigation of the anatomy of its main activities (A31, A32). For the activity of 'Annual Cropping' (A31) the sectors of Irrigation, Harvesting and Fertilising are identified. Seasonality, crop rotation and adopted strategies control the environmental and economic outputs of each of the sectors. The notion of 'virtual water' is introduced in this diagram to depict the water flow embedded in the agricultural products. As discussed earlier, several of the environmental and economic inputs and outputs of the sectors are considered externalities to the catchment system. A combination of physical and natural mechanisms with the involvement of the human capital result in the delivery of the products and services. Similarly, for the activity of 'Farming' or 'Livestock' (A32) the sectors of watering and feeding the herd are identified. The same principles and assumptions regarding the externalities, the controls and the mechanisms apply for this activity of the actor 'Agriculture'. The diagrams A31 (**Figure 5.11.**) and A32 (**Figure 5.12.**) constitute the ground for the creation of the Catchment PIOT Agriculture section.

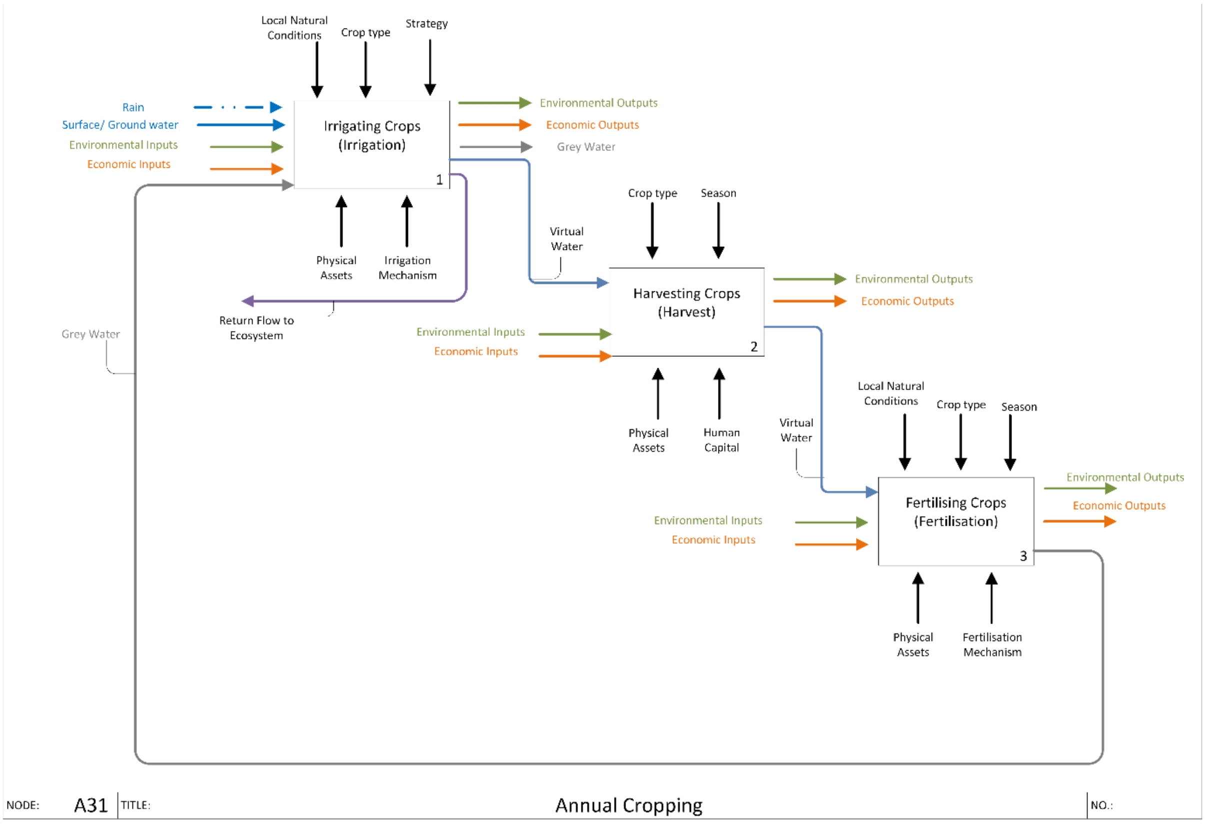


Figure 5.11.: The analysis of the agricultural subsystem 'Annual Cropping' (A31 IDEF0 diagram). The level of complexity in the diagrams increases as a number of water flows are embedded in economic inputs and outputs.

The scope and level of analysis of each of the subsystems of a catchment system will lead to the further development of the IDEF0 model and the construction of more IDEF0 diagrams. The granularity of the model is very much dependent on the scope of the analysis, and on data availability. The IDEF0 model created for the selected catchment system leads to the creation of the Catchment PIOT. In its final format, the Catchment PIOT is a matrix of flows, both physical and economic, circulating within the catchment boundaries. To achieve this format, the cells of the Catchment PIOT are filled in using indexes from Water Accounting techniques, where the output of each of the sectors (row) to the other sectors (column) are depicted. As a result, each column represents the figures related to the inputs received by a single metabolic compartment of the system. Similarly, to the original PIOT, this procedure assists to the visualisation of the quantitative information relating to each component ('sector') of the catchment in the form of inter-component exchanges. The indexes for flow accounting and estimation of environmental outputs generate the 'Appendix' of the Catchment PIOT and the computations performed as part of it feed into the Catchment PIOT cells. They will be extensively discussed in the following chapter (Chapter 6).

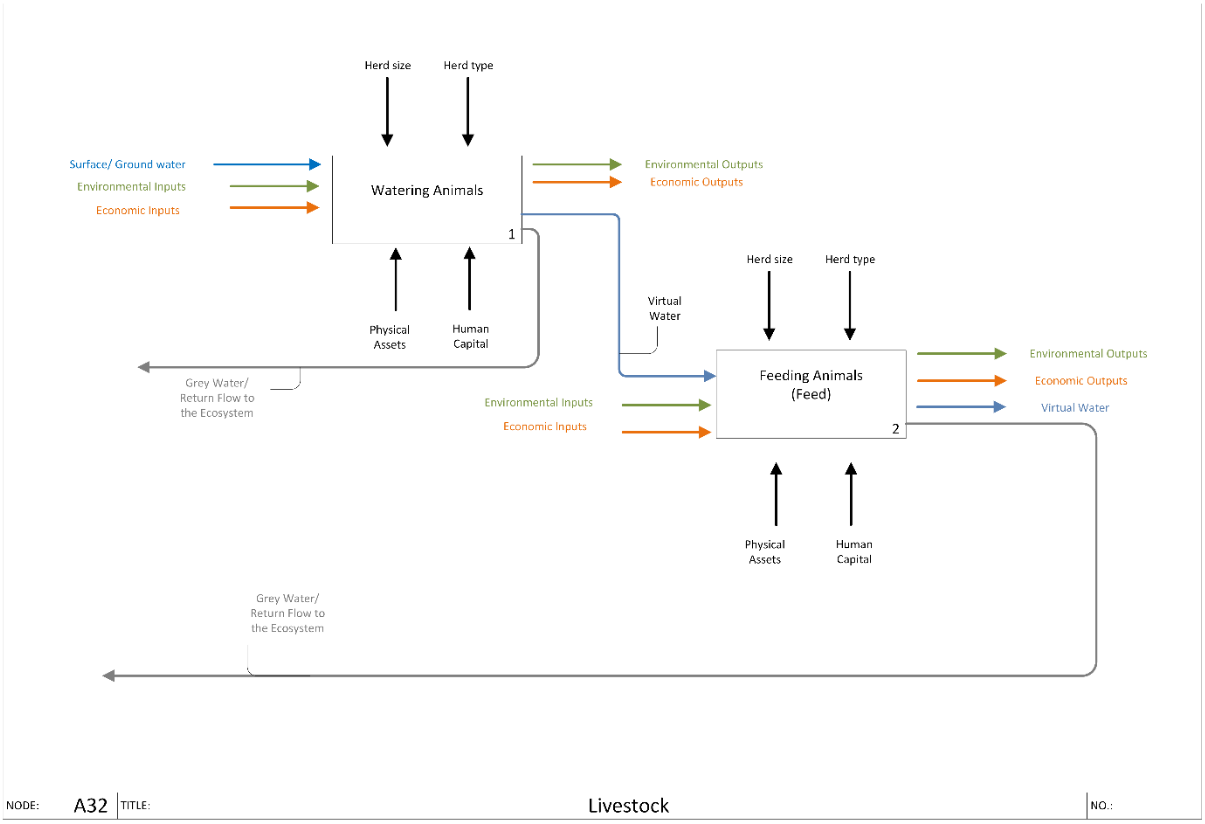


Figure 5.12.: The analysis of the agricultural subsystem 'Livestock' (A32 IDEF0 diagram). Embedded water flows (virtual water) increase the complexity of the diagram. The linkages with the subsystem 'Annual Cropping' are not depicted; but can be added as dictated from the research needs or allows by the level of available data.

After describing the rules and principles underpinning the creation of the Catchment Metabolism modelling schema, the chapter continues by presenting its construction. Its building blocks are concrete steps (**Figure 5.13.**) that synthesise a new approach to asset management and to the representation of catchment systems. The first steps include the definition of the scope of the catchment analysis and the identification of the water actors/industries of the systems, whose activities are relevant to the scope. Then the industries are broken down into the sectors they consist of and an initial sector by sector matrix can be formulated in order to facilitate the next steps. The IDEF0 diagrams and model are constructed in the following phase, with the expert input of several specialists. The Catchment Physical Input Output Table can then be created, based on the outcomes of the IDEF0 catchment model. The design of the catchment system dynamics follows, where the interlinkages among actors are depicted. In parallel, the accounting appendices are formulated which show the indexes and algebraic equations used for the computations of the final outputs for the Catchment PIOT. Despite the pre-defined steps and phases, the application of the Catchment Metabolism modelling schema is a rather iterative process. The actions undertaken need to map the original scope, while the outcomes of each of the steps must feed into the following phase.

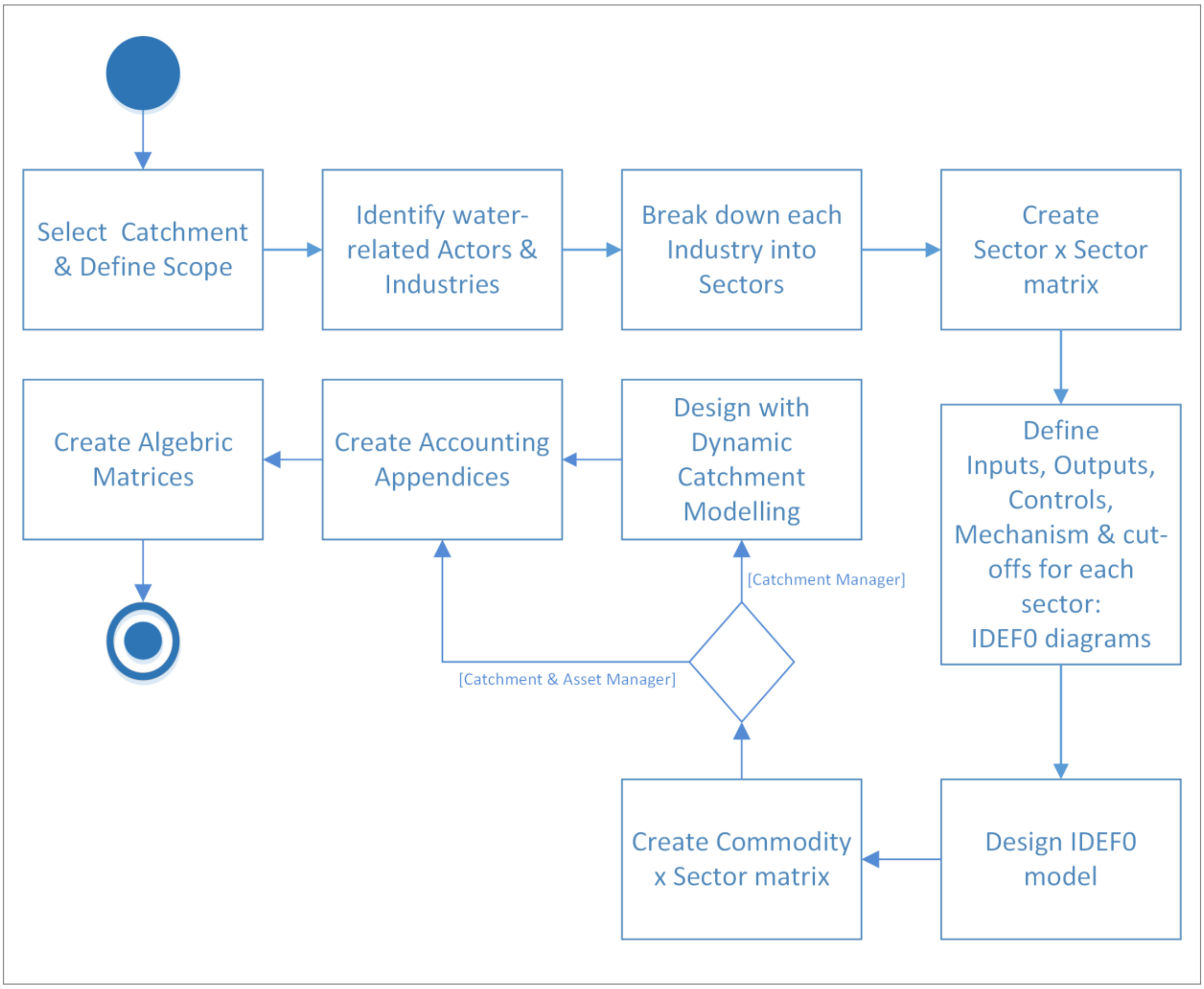


Figure 5.13.: The steps undertaken to produce the Catchment Metabolism modelling schema for a selected catchment system.

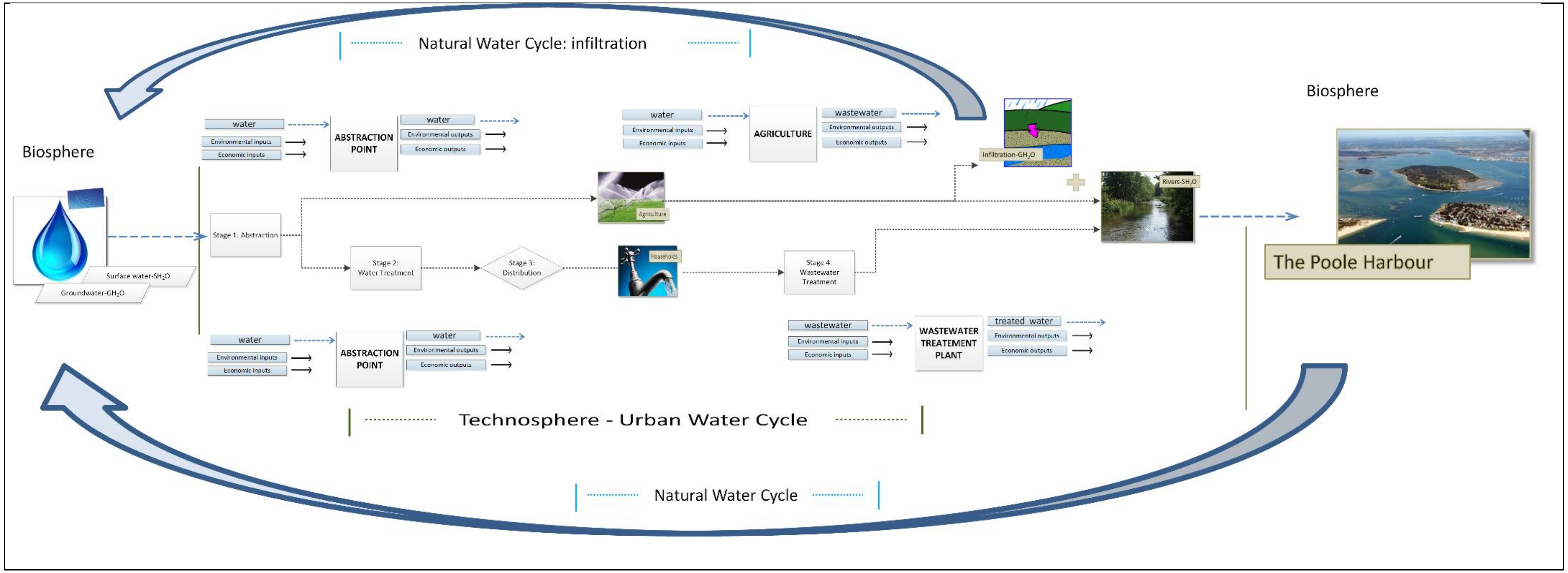


Figure 5.14.: The Catchment as a System. Modified flow chart depicting the water actors and flows within the Poole Harbour catchment.

Table 5.1.: The Catchment Physical Input Output Table (Catchment PIOT). The values within the cells are indicative of outputs generated by activities of the water actors of the catchment. The outputs mainly describe volumes of water (m^3) or the metrics from indicators (non-dimensional).

		Ecosystem Functions				Water Services					Agriculture					
		Water Cycle				Urban Water Cycle					Annual Cropping			Livestock		
		Atmosphere	Hydrosphere	Pedosphere	Lithosphere	Abstraction	Water Treatment	Water Distribution	Wastewater Distribution	Wastewater Treatment	Irrigation	Harvest	Fertilising	Watering Animals	Feed	
		1	2	3	4	5	6	7	8	9	10	11	12	13	14	
Atmosphere	1	X (1,1)														X (1,n)
Hydrosphere	2		X (2,2)								X (2,10)					X (2,n)
Pedosphere	3			X (3,3)												X (3,n)
Lithosphere	4				X (4,4)	X (4,5)					X (4,10)			X (4,13)		X (4,n)
Abstraction	5	X (5,1)				X (5,5)					X (5,10)			X (5,13)		X (5,n)
Water Treatment	6	X (6,1)					X (6,6)									X (6,n)
Water Distribution	7							X (7,7)			X (7,10)			X (7,13)		X (7,n)
Wastewater Distribution	8								X (8,8)							X (8,n)
Wastewater Treatment	9	X (9,1)								X (9,9)						X (9,n)
Irrigation	10		X (10,2)	X (10,3)	X (10,4)						X (10,10)					X (10,n)
Harvest	11											X (11,11)				X (11,n)
Fertilising	12												X (12,12)			X (12,n)
Watering Animals	13		X (13,2)	X (13,3)	X (13,4)									X (13,13)		X (13,n)
Feed	14	X (14,1)	X (14,1)												X (14,14)	X (14,n)
		X (n,1)	X (n,2)	X (n,3)	X (n,4)	X (n,5)	X (n,6)	X (n,7)	X (n,8)	X (n,9)	X (n,10)	X (n,11)	X (n,12)	X (n,13)	X (n,14)	

5.3.2. The Catchment Metabolism schema in a water company

Applying the Catchment Metabolism modelling schema in practice requires the input from a number of experts, as for the transdisciplinary nature of the approach calls for the synthesis of a wide spectrum of expertise. The modified use case diagram (**Figure 5.15.**) demonstrates the types of experts and their individual contributions for the design and application of the Catchment Metabolism schema.

The use case diagrams are a software engineering technique and are used as a simple, but powerful tool to express the functional requirements of a system (Papajorgji and Pardalos 2014). Their construction is based on the object-oriented approach, such as the Unified Modelling Language (UML). A use case diagram contains information about the actors (i.e. the users of the system) and about the use cases (i.e. what the users do within the system). For the modified use case presented here, the “object” around which the diagram is drawn is the Catchment Metabolism modelling schema and the “system” is a water company.

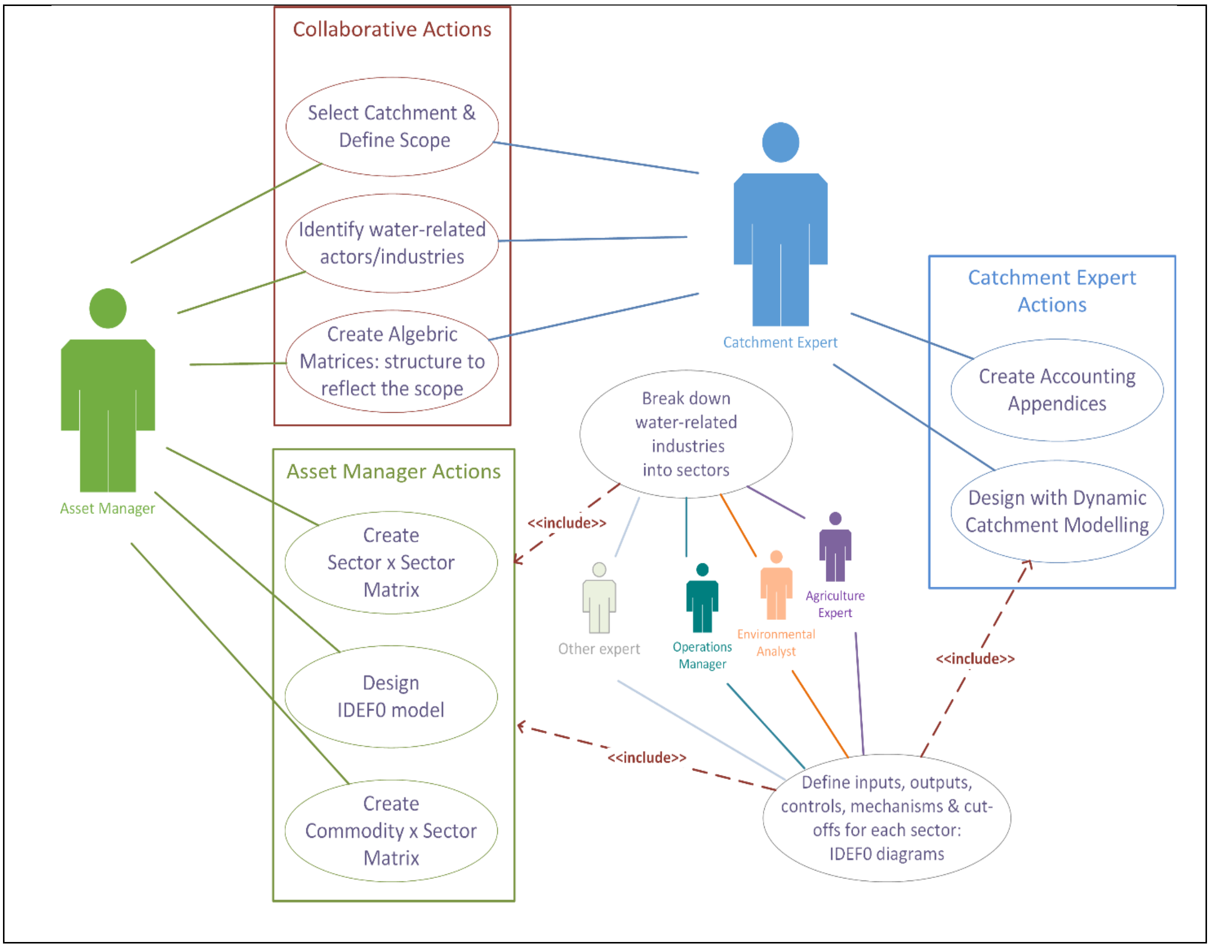


Figure 5.15.: Modified use case diagram on the expert input for the production and implementation of the catchment metabolism schema within a water company. The Unified Modelling Language (UML) is the basis for the construction of this diagram; derogations from the UML rules were made for the accommodation of the scope of the work.

The main actors identified in the given system are an Asset Manager and a Catchment Expert. Throughout the process these two actors are heavily involved. These roles can be fulfilled by individuals or teams. Their common tasks include the definition of the scope of the application and the identification of the main water actors in the catchment, i.e. of the catchment metabolism. Their individual tasks reflect their particular skills knowledge and are also aligned with the input from other company or external experts. For their individual tasks, the Asset Manager is responsible for the construction of the matrices that represent the outputs of individual sectors or activities within the catchment boundaries, while the Catchment Expert develops the accounting mechanisms for the computations of the outputs, making use of water accounting techniques.

However, the practical application of the schema is a rather comprehensive process which requires the input from multiple experts and collaborative action to be taken. For the creation of the Catchment PIOT and the IDEF0 model, a number of experts are required in order to perform the break-down of the water-related industries into their sectors and define their structural features (inputs, outputs, controls, mechanisms) respectively. For the case study analysed in this work, the expertise of an environmental analyst, an operations manager and an agricultural expert are required for the analysis of the building blocks of the three main water actors identified within the given catchment system.

The data produced by the assembly of the separate IDEF0 diagrams constitute the heart of the entire schema providing essential insights in the subsystems of the catchment under consideration. The Asset Manager will then pull the separate IDEF0 diagrams together in order to create the IDEF0 model and the Input-Output matrices for sectors and commodities. The data gathered for the development of the IDEF0 model will serve as the basis for the construction of a systems dynamic model by the Catchment Expert. The outputs of this type of model will produce the information for the Catchment PIOT.

5.4. Summing up the Catchment Metabolism modelling schema

The research outputs described in this chapter provide a novel, structured and systemic approach for asset management schemes in the water sector. The approach enables the integration of natural assets in the water sector’s portfolio and contributes to the limited literature of the approaches on transparent flow accounting and industrial reporting.

The Catchment Metabolism is a modelling schema built on an interdisciplinary basis. The building blocks of the underpinning methodology have been analysed and introduced via a selected case study. The well-defined structure of the creation of the modelling schema provides an opportunity for standardising an approach which allows water companies to explicitly account for natural capital and respond to current policy demands for resilient and long-term investment planning. The application of IDEF0 logic and rules for performing a catchment analysis provides consistency in modelling different and diverse systems.

After having introduced the conceptual part of the Catchment Metabolism modelling schema and the principles of its underpinning methodology, the next chapters will focus on the synthesis of a set of metrics and indexes that allow for effective catchment flow accounting. The outputs of the processes presented in the IDEF0 model will be computed and used to convert the Catchment Physical Input Output Table into a portfolio representation on process outputs.

Chapter 6: The Catchment Metabolism in action: creating the Water Inventories

Chapter 6 presents the application of the Catchment Metabolism in the selected case study (the Poole Harbour Catchment) and demonstrates the process of computing the content of a Catchment Physical Input-Output Table (Catchment PIOT). The Water Inventories underpinning Catchment PIOT were created based on the research advancements of the field of Water Accounting. Actor-specific water accounts are produced for each of the actors identified in the stakeholder analysis performed previously. The methodological choices made and data used for the arithmetic calculations of the hydrological water balance of the catchment (ecosystem water account) and the water inventory and environmental outputs (water company, agriculture) are presented and discussed.

For the identification of the causal relationships among the catchment's stakeholders, a systems dynamics analysis is conducted. A scenario analysis and assessment is then performed, based on the methodologies created for each of the catchment's stakeholders. The scenarios describe the current environmental status of the catchment and assess how two alternative approaches to agricultural practice will affect the overall environmental performance of the catchment.

The research outputs of the chapter set the rules for the creation of the Catchment PIOT Water Inventories based on the established methods and indices of Water Footprint Assessment and Life Cycle Assessment. They contribute transparent methodologies for stakeholder-specific water inventories, applied at a catchment scale. These map hydrological parameters against the available indicators provided in literature and demonstrate their use within the context of Integrated Catchment Management, contributing to the limited relevant literature. Further, the synthesis of methods and approaches explored in the chapter, i.e. systems dynamics and water footprinting, indicates a novel perspective on their combined use.

6.1. Methods Review

In the field of Water Accounting, literature shows two parallel developments in the area of Water Inventories: water-related Life Cycle Assessment (LCA_{water}) and Water Footprint Assessment (WFA). Boulay et al. (2013) summarise, compare and contrast the two methodologies (**Table 6.1.**).

Briefly, both methodologies comprise of four stages, namely: scope definition, inventory or accounting, impact or sustainability assessment and interpretation or response formulation. Further, both methodologies comply with the ISO standard 14046 (2014) requirements for water footprinting, as they include water consumption and pollution and can be used in synergy with other water management tools to meet their goal: enable practitioners to preserve water resources (Pfister and Ridoutt 2014; Boulay et al. 2013).

Herein, the short introduction to the methodologies is followed by the more detailed presentation and discussion of the indices used for the performance of the water-related environmental impact assessments. The academic literature shows that the research area of water inventories is rapidly growing and lively discussed. The debate between the LCA_{water} and the WFA communities has been more active since the publication of the BS ISO Standard on Life Cycle Assessment and Water Footprint (BS ISO 14046:2014), with an increasing number of publications focussing on new methods for assessing environmental impacts (e.g. Boulay et al. 2015a,b,c; Pfister and Bayer 2014) and critical analyses and review (e.g. Berger et al. 2016; Núñez et al. 2016; Hoekstra 2016).

Table 6.1.: Comparison and synergies between the Water Footprint Assessment (Hoekstra et al. 2011) and the water-specific Life Cycle Assessment (LCA_{water}) methodologies. Based on Boulay et al. 2013.

Synergies between LCA_{water} and Water Footprint Assessment (WFA) methodologies	
LCA_{water}	**WFA**
Inventory -The quantitative indicators from WFA can be used within the LCA inventory, particularly the blue WF. -The use of Green WF is restricted, due to the lack of green water use pathways. -The use grey WF is not advised, due to the hypothetical quantification of water pollution.	**Water Footprint Accounting** -LCA inventory data can be obtained from the well-developed existing databases.
Impact Assessment -The WFA Blue Water Scarcity indicator can be compared with other water scarcity indicators from LCA.	**Sustainability Assessment** -LCA impact assessments can be considered in WFA to better evaluate the sustainability of fresh water consumption.
Interpretation -The sustainability assessment and response formulation from WFA can be used to improve the interpretation of LCA quantitative results.	**Response formulation** -No synergies identified.

This section intends to give an overview of the developments in the field of water inventory and of the predominant indices used in the literature of both communities. The synergies, differences and limitations of the existing methods are discussed, aimed at identifying the indicators suitable to serve the needs of the implementation of the Catchment Metabolism modelling schema in practice.

6.1.1. Inventory or Accounting Phase

The ISO standard 14046 (2014) defines the life cycle inventory analysis (or LCI) as the phase of life cycle water footprint assessment involving the compilation and quantification of input and outputs related to water for a product throughout its life cycle. The development of water LCI schemes has progressed over the years, as evidenced by the growing number of publications and modifications over the years (**Figure 6.1.**).

Owens (2002) introduced the terminology used in the water-related LCA advancements and set the grounds for the development of LCI water schemes. The LCI schemes developed by Vince (2007), Bayart et al. (2010) and Boulay et al. (2011) propose a detailed accounting of water use which considers volumetric, geographical, watercourse and quality information in order to satisfy the requirements of the recently developed impact assessment methods (Berger et al. 2016). Pfister et al. (2015) graphically summarise the inventory flows relevant to the assessment of water use impacts, as they have been incorporated in the Ecoinvent 3.0 database (**Figure 6.2.**).

The flow accounting is performed according to the principles outlined in the earlier works (Bayart et al. 2010; Owens 2002) which suggested that, two parameters need to be considered for each water flow: 1. Resource type (e.g. groundwater, surface water), 2. Water quality. These works also highlighted the need for the calculation of the balances of each of the elementary water flows of the system as a means to quantify the net changes of the availability of each of them.

The term 'water use' in LCI schemes is defined as the total withdrawal of freshwater (Berger et al. 2016; Pfister et al. 2009), which is differentiated into: consumptive water use (i.e. water flow not returned in the original water basin due to evapotranspiration, product integration or discharge into other basins or the sea), degradative water use (i.e. water flow returned to the original water basin, after qualitative degradation) and borrowing water use (i.e. water flow withdrawn and discharged with no or low quality degradation. The definition of the consumptive water use is challenged and criticised by Burger et al. (2014), which opposes that the LCI definition neglects the significant shares of evaporated water returning via precipitation within short time and length scales. The work introduces the concept of effective water consumption (WC_{eff}) and the relevant accounting scheme, which considers the effects of atmospheric moisture recycling within basins. The introduction of evaporation cycle process in the water accounting process of products or services is mostly relevant when considering agricultural product systems. Further, in the existing LCI schemes, “green water” (i.e. the part of precipitation stored in the soil or the precipitation that temporarily stays on top of the soil or vegetation) is not included in the consumptive water use, as it is considered as a land use indicator, not fully affecting the water cycle (Ridoutt and Pfister 2013). The non-inclusion of green water in LCI schemes is also reflected in the identification of the potential impact pathways (**Figure 6.3.**).

The framework proposed by Bayart et al. (2010) suggests that water quality in the LCI modelling can be considered using two distinct approaches: (i) distance-to-target or (ii) functionality**.** For the former, the quality of the different water types is assessed by determining the equivalent effort necessary to process each water source to the same final effluent. This can be done by assessing either the volume of water needed to dilute a given water type to the acceptable standards for each single use or the energy required to purify a resource at the same quality. For the latter, a water flow is functional when its quality parameters meet the acceptable standards concerning each user. In this vein, Boulay et al. (2011) introduced 17 distinct water categories based on the source, quality and potential users as a means to assess the loss of functionality for humans. Recently, Berger et al. (2016) stress that, further methodological developments are needed to include water quality parameters into the corresponding water impacts.

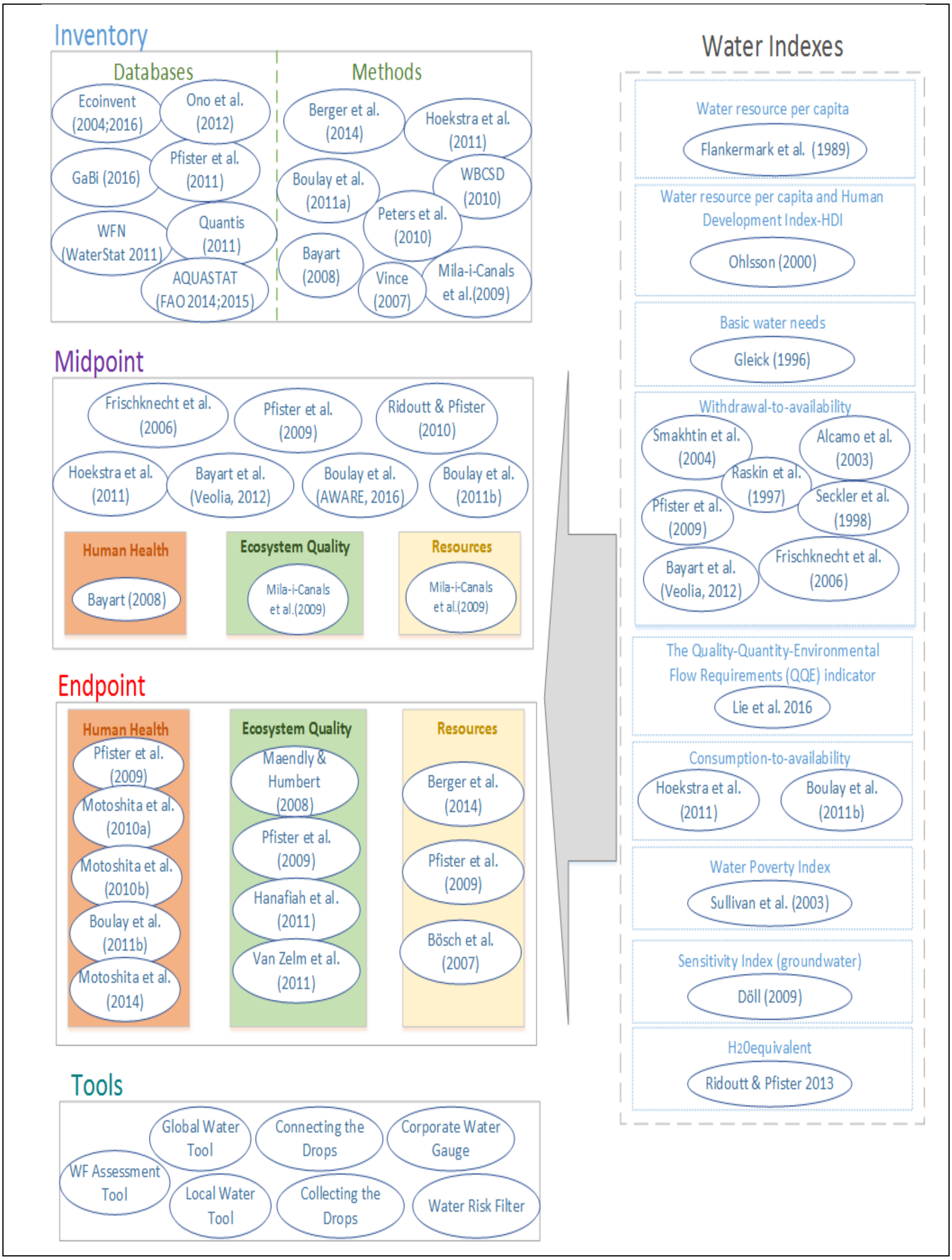


Figure 6.1.: Available freshwater inventory and impact assessment methods and water indices. Adapted from Kounina et al. 2013 and enhanced based on more recent literature.

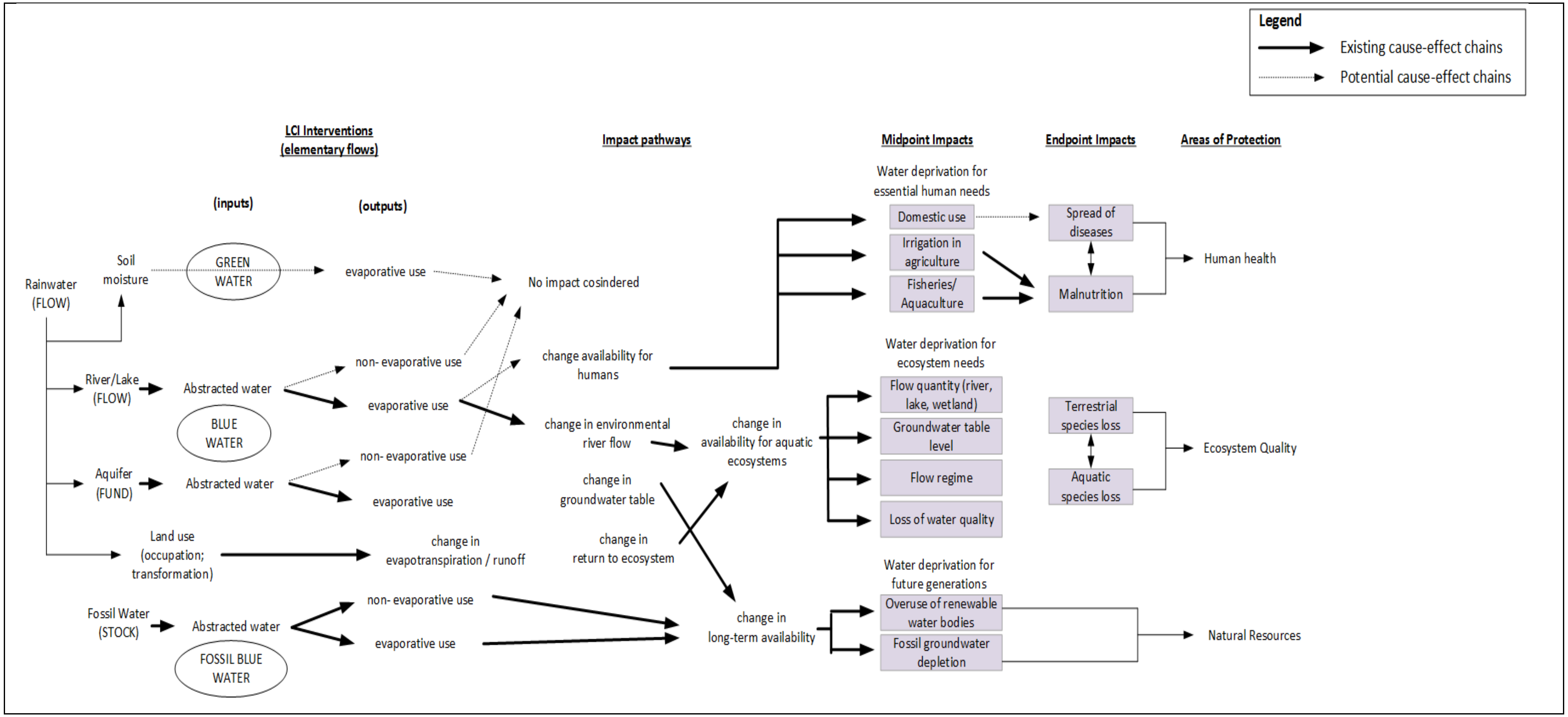


Figure 6.3.: Main Impact Pathways relating to freshwater use in Life Cycle Assessment studies. Adapted from Mila i Canals (2009) and updated based on Kounina et al. 2013.

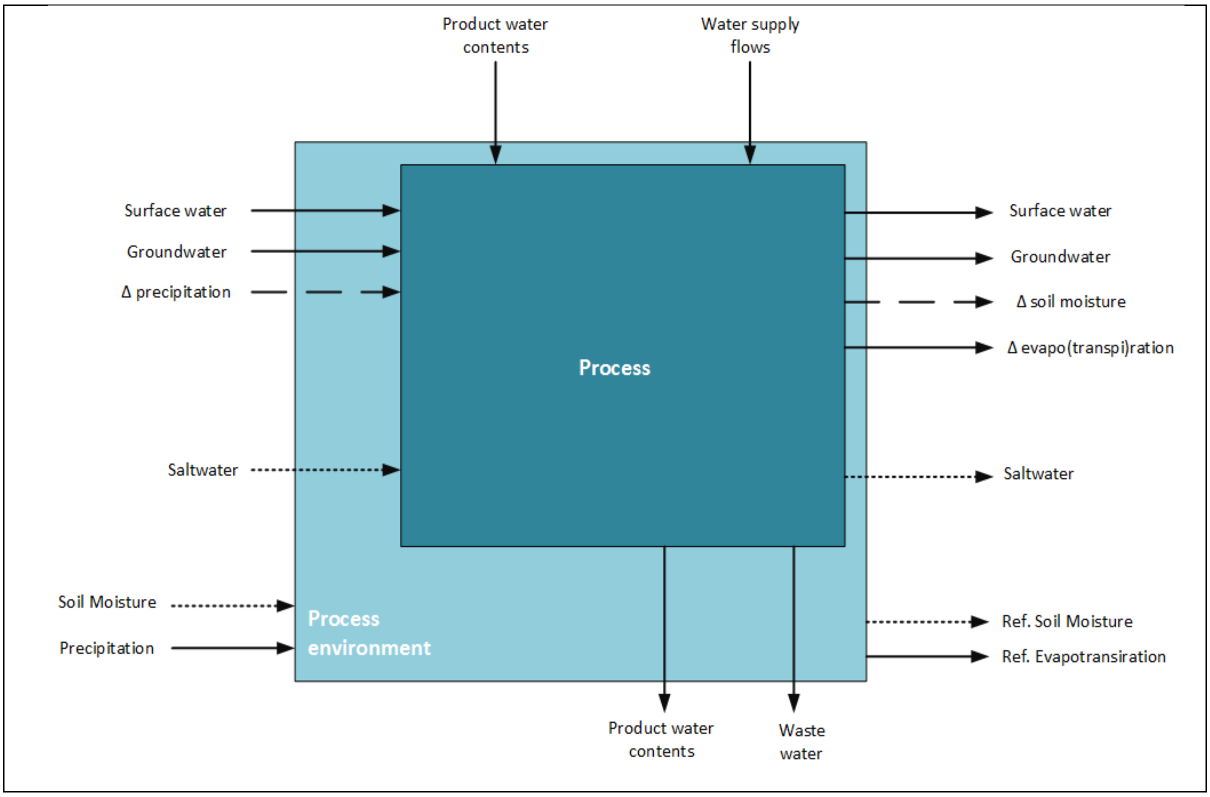


Figure 6.2.: The inventory flows relevant to the assessment of water use impacts. Adapted from Berger et al. 2016 and Pfister et al. 2015.

For the Water Footprint Assessment (WFA) methodology, the equivalent of the LCI scheme is the Water Footprint Accounting phase. This includes the identification of the inventory boundaries (i.e. system boundaries, type of footprint considered, spatiotemporal explication) and the consequent computation of the water footprints (Hoekstra et al. 2011).

The water footprint (WF) is an indicator of freshwater use which includes both the direct and indirect use of water of a consumer or a producer (Hoekstra, 2003, **Figure 6.4.**) and its development was based on the concept of virtual water as introduced by Allan (2003). It is a volumetric, multidimensional indicator, showing water consumption volumes by source and polluted volumes by type of pollution. All components of a total water footprint are specified geographically and temporally: the WF is a geographically and temporally explicit indicator, showing not only the volume of the consumptive water use and pollution, but also the locations and time. The WF can be calculated for different entities (e.g. a step, a process, a product, a nation etc.), different groups of consumers (e.g. an individual family) or producers (e.g. an enterprise or an economic sector) and for different geographically delineated spatial scales (e.g. a country, a region or a catchment). For example, the WF of a product is the volume of freshwater used to produce the product, measured over the full supply chain and the WF of an individual, community or nation is defined as the total volume of freshwater that is used to produce the goods and services consumed by the individual, the community or the business.

As shown in **Figure 6.5.,** a WF consists of three components: blue WF (WF_{blue}), green WF (WF_{green}) and grey WF (WF_{grey}). The WF_{blue} measures the consumptive use of fresh surface and/or groundwater, the so-called blue water. The term ‘consumptive water use’ refers to one of the following four cases: (1) water evaporates, (2) water is incorporated into a product, (3) water does not return to the same catchment area, (4) water does not return in the same period. The WF_{blue} differs from ‘water withdrawal’ in three main points: (1) it does not include blue water use that is returned to where it came from (2) it considers blue, green and grey water (3) it includes both direct and indirect water use. The WF_{green} quantifies the human consumption of the green water (i.e. the part of precipitation stored in the soil or the precipitation that temporarily stays on top of the soil or vegetation) and is particularly relevant for agricultural and forestry products (products based on crops or wood). It refers to the total rainwater evapotranspiration (from field and plants) plus the water incorporated into the harvested crop or wood. The WF_{grey} indicates the volume of freshwater that is required to assimilate the load of pollutants based on the natural background concentrations and existing ambient quality standards.

According to the WFA manual (Hoekstra et al. 2011), the tool of WF is not a measure of the severity of the local environmental impact of water consumption and pollution; therefore, it does not address environmental issues other than freshwater scarcity and pollution. Thus, it should be regarded as an analytical tool that has broadened the traditional scope in water scarcity analysis by introducing supply chain thinking and can expand the knowledge base for integrated water resources management (IWRM) or integrated catchment management (ICM) and informed decisions being made in these contexts. In such cases, the WF is used to express human appropriation of freshwater in volume terms, when compared with the hydrological cycle within a river basin or catchment. **Figure 6.6.** shows the green and the blue WF in relation to the water balance of a catchment or river basin. For the computation of the total WF of a delineated area or a catchment, the sum of the total freshwater consumption and pollution within the boundaries of the area needs to be calculated (**Figure 6.7.**) It is noted that, in order to calculate the total catchment WF, various water balances and footprints need to be computed, including the WF of consumers living within the catchment and the balance of the virtual water flows (i.e. water incorporated in the products) which are imported and exported to and from the catchment.

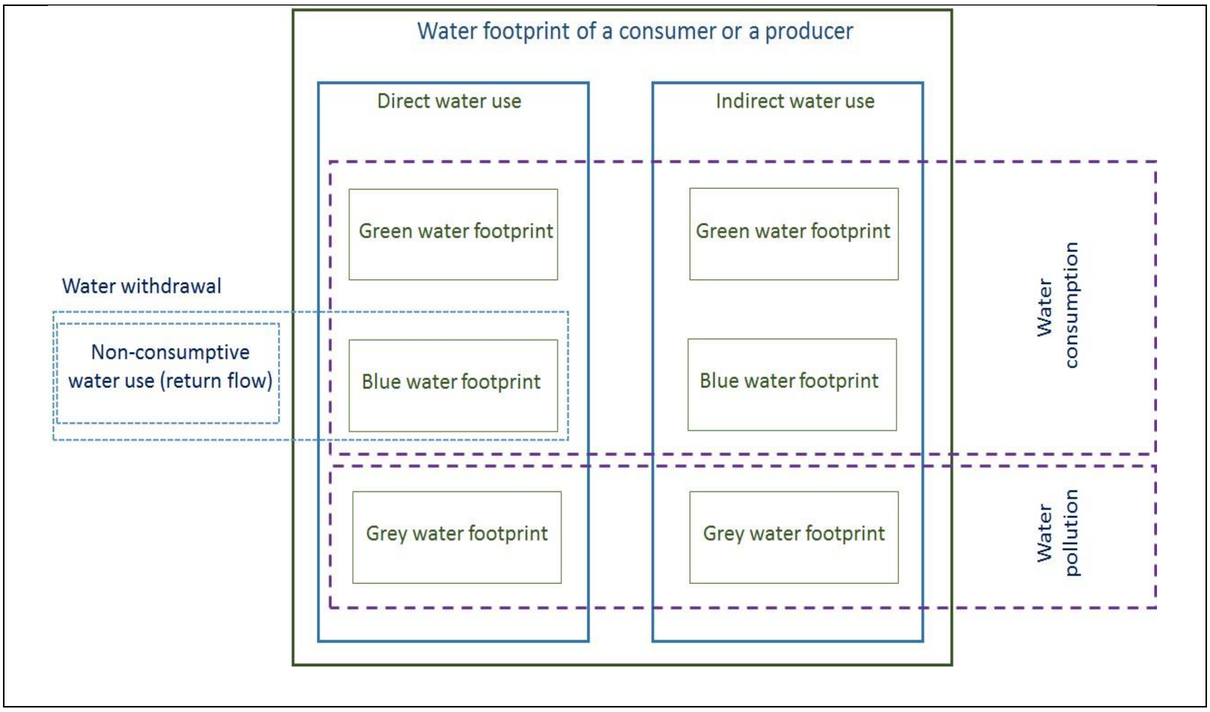


Figure 6.4.: Schematic representation of the components of a water footprint. It shows that the non-consumptive part of water withdrawals (the return flow) is not part of the water footprint. It also shows that, contrary to the measure of 'water withdrawal', the 'water footprint' includes green and grey water and indirect water-use component. Adapted from Hoekstra et al. 2011.

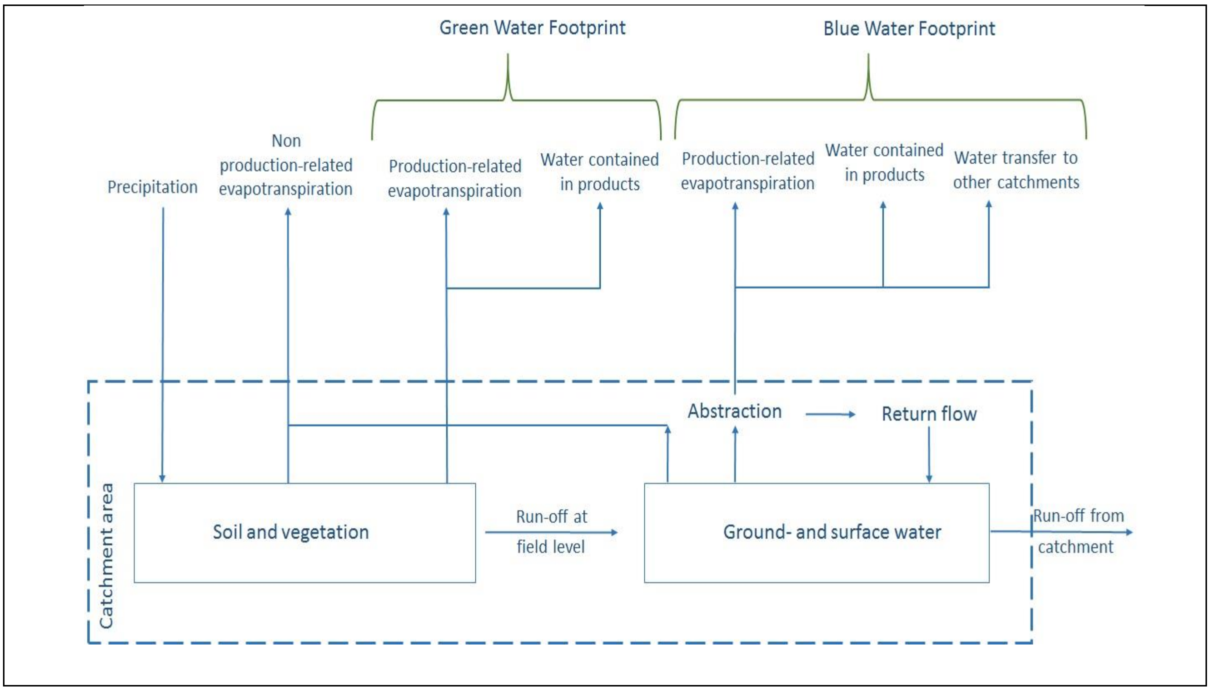


Figure 6.5.: The green and the blue water footprint in relation to the water balance of a catchment area. Adapted from Hoekstra et al. 2011.

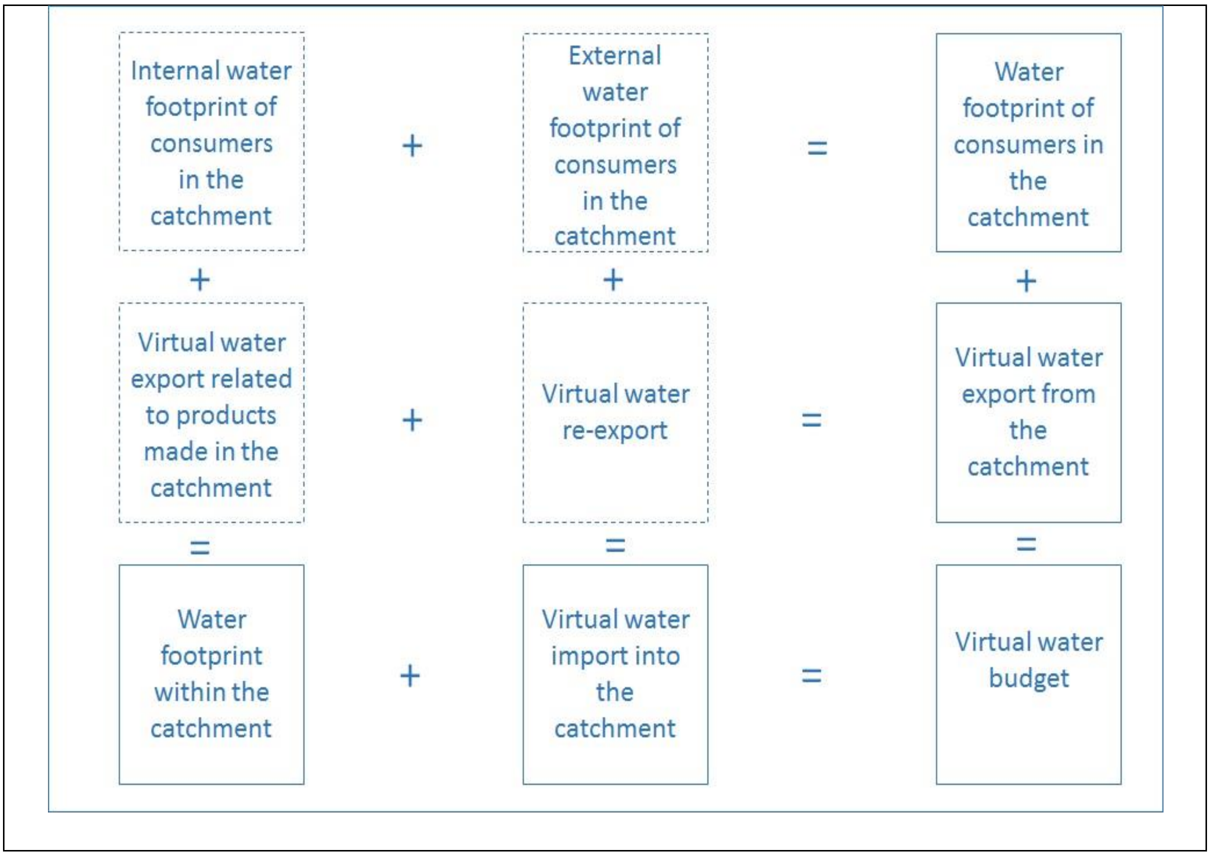


Figure 6.6.: The catchment water footprint (wf) accounting scheme. It shows the various balances that hold for the wf of consumers living within the catchment, the water footprint within the catchment area, the total virtual water export from the catchment and the total virtual water import into the catchment.

The concept of WF and the WFA methodology have been broadly accepted by global and national policy-makers, including the United Nations Environment Programme (UNEP) and the Environment Agency (EA) of the United Kingdom, and have been applied to a number of projects, such as the umbrella project 'Water Footprint Neutrality & Efficiency (WaFNE)' (UNEP 2011) and the 'Water Footprint Assessment for the Hertfordshire and North London Area' (Zhang et al. 2014). The outputs of these projects have highlighted areas of improvement in different research areas and economic sectors or even formulated responses for specific regions. In the research arena, the WFA methodology has been widely applied. Nevertheless, the WFA methodology and its indicators have been largely criticised Wichelns 2015; Chenoweth et al. 2014; Ridoutt and Pfister 2013; Yang et al. 2013).

6.1.2. Impact or Sustainability Assessment Phase

Water use is addressed twice in the Life Cycle Assessment (LCA) compendium: as an impact category within LCA (ISO 14040:2006) or as a stand-alone 'water footprint' (ISO 14046:2014), which is defined as a "metric that quantifies the potential environmental impacts related to water". According to the recommendations of the WULCA group (http://www.wulca-waterlca.org/), these include impacts related with water use, and the subsequent effect on water availability for humans and ecosystems as well as direct impacts on the water resource and its users from emissions to air, soil and water. The traditional LCA impact categories (e.g. freshwater eutrophication, freshwater acidification etc.) are used for the quantification of impacts. A water footprint can be represented as a result of a stand-alone assessment or as a sub-set of results of a larger environmental assessment, such as a full-LCA (**Figure 6.7.**). According to the ISO standard 14046:2014, the term 'Water Footprint' is applied only when both consumptive and degradative aspects of water use are assessed. When the indicators are used within more specific contexts, their name should reflect the scope. For example, when only consumptive water use is assessed, the 'water scarcity foorprint' should be used as an alternative.

The developments in the LCA impact assessment frameworks and methods have flourished during the last few years, as witnessed by the considerable volume of published literature (**Figure 6.1.**). The ongoing developments on the assessment of freshwater use have driven the publication of a number of review papers (Berger et al. 2016; Núñez et al. 2016; Boulay et al. 2015a,b; Jarvis et al. 2013; Kounina et al. 2013; Berger et al. 2013; Jeswani et al. 2011; Berger et al., 2010) and case studies (van Hoof et al. 2013; Godskesen et al. 2013, Uche et al. 2013; Yang et al., 2013, Angrill et al. 2012; Gleeson et al. 2012; Jefferies et al. 2012; Stoeglehner et al. 2011; Pfister et al. 2009), which address regional water resources at various scales (product, aquifer, hinterland, urban, groundwater catchment, watershed).

On the midpoint level, the basic and common concept of indicators developed to assess the environmental impacts of freshwater use is to express the physical resource availability compared to the demand by taking the ratio of water use or consumption to water availability. As discussed by Boulay et al. (2015a), the indicators have evolved from Withdrawal-to-Availability (WTA) and Consumption-to-Availability (CTA) to Demand-to-Availability (DTA) and Availability-minus-Demand (AMD). Based on the latter, the AWARE (Available Water Remaining) method (Boulay et al. 2016, submitted) has been suggested as a generic midpoint indicator for assessing water consumption. The indicator represents the relative available water remaining per area in a watershed (expressed in m^3 m^{-2} $month^{-1}$), after the demand of humans and ecosystems have been met. Thus, it assesses the potential of water deprivation of either humans or the aquatic ecosystem and can be classified as a water scarcity footprint indicator.

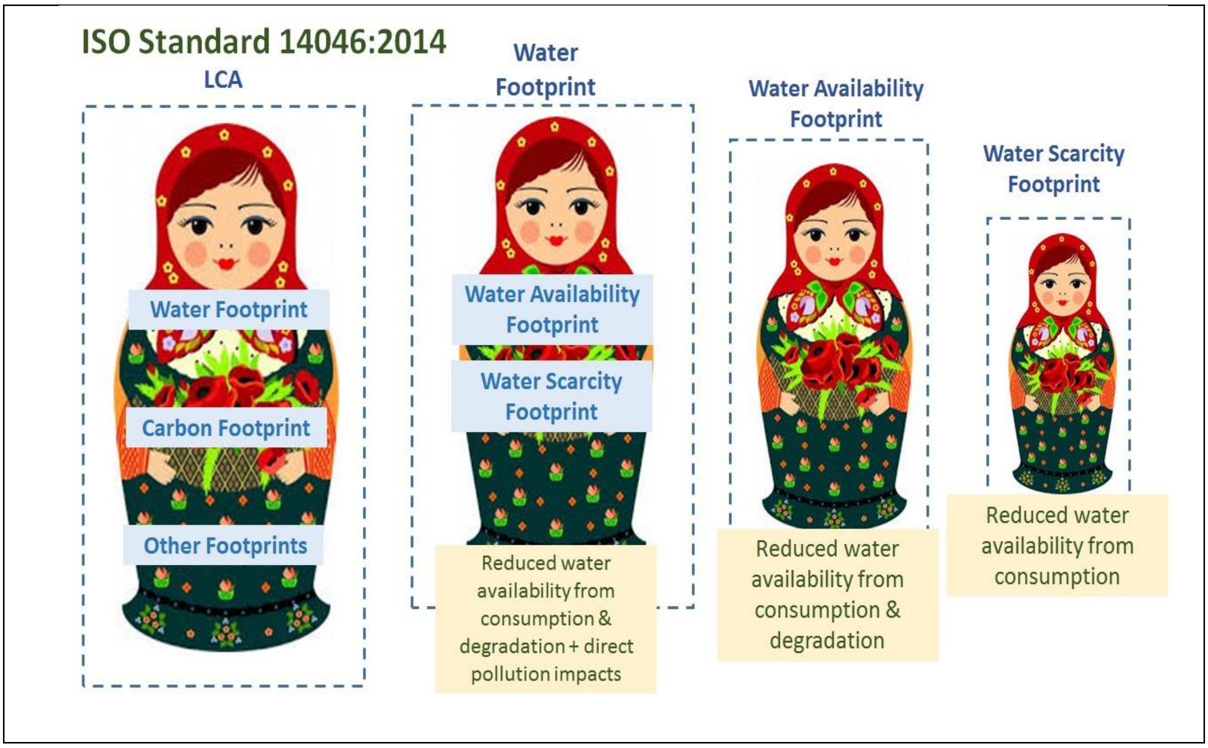


Figure 6.7.: The different types of Water Footprint according to the ISO standard 14046:2014.

On the endpoint level, the water use can result in three areas of protection (Kounina et al. 2013), relating to "human health", "ecosystem quality" and "resources" (**Figure 6.1.**; **Figure 6.3.**). The cause-effect chain on human health is relatively well-defined compared to the other two categories. Recent works (e.g. Boulay et al. 2015 a,b,c) identify the key existing methods and discuss methodological developments for achieving consensus for the assessment of endpoint impacts relating to human health due to water consumption or scarcity. The impact pathways of water consumption to ecosystem quality and to resource availability are more complicated and the targets and approaches for assessing potential damages are diverse (Berger et al. 2016). There are several limitations for the expansion and consensual adoption of the available methods (**Figure 6.1.**) due to lack of knowledge of region-specific pathways and data availability of parameters for the analysis. Considerable differences and relevant limitations apply to the characterisation factors used in LCA studies, while the use of water indices as characterisation factors for midpoint and endpoint impact assessments methods has been suggested (Kounina et al. 2013) for studies on freshwater consumptive or degradative use.

The counterpart of the 'Impact Assessment' phase for the Water Footprint Assessment (WFA) methodology is referred as 'Water Footprint Sustainability Assessment'. This phase assesses the sustainability of water footprints when compared with the water availability at a given unit of analysis (Hoekstra et al. 2011). The assessment is based on the principle of 'environmental sustainability boundaries' (Richter 2010) and the concept of primary and secondary impacts (i.e. the equivalent of midpoint and endpoint impact categories of LCA studies respectively).

When conducting a water footprint sustainability assessment of a catchment or a river basin, a four-step process is followed (**Figure 6.8.**). The first step involves the identification and quantification of the criteria for conducting the sustainability assessment of the catchment. The identification of the catchment's hotspots, namely sub-catchments or periods of the year when the water footprint is regarded unsustainable, follows. The third and fourth steps involve, respectively, the quantification of the primary and secondary impacts in the hotspots. Primary impacts are described in terms of changed water flows and quality (compared to the natural conditions, without human disturbances). Secondary impacts refer to the ecological, social and economic goods and services that are impaired from the catchment area as a result of the primary impacts.

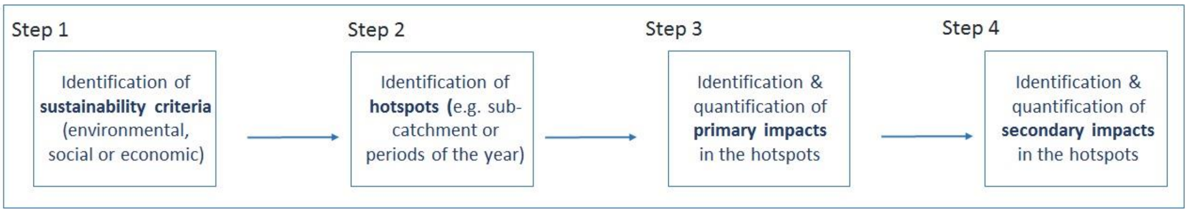


Figure 6.8.: Assessment of the sustainability of the water footprint within a catchment or river basin as a step-four process (adapter from Hoekstra et al. 2011).

The identification of the catchment's environmental hotspots is a substantial step in a Water Footprint Sustainability Assessment. The environmental hotspots can relate to the green, blue or grey footprint of the catchment and a set of mathematical figures are used for their identification. A number of indices has been introduced in the literature (Hoekstra et al. 2011) and are summarised in **Table 6.2.** In general terms, the water (green or blue) availability of the catchment is firstly computed. The figures produced are then used to produce the water (green or blue) scarcity figures. The water scarcity indicators denote the 'fraction of appropriation' of the available freshwater resources. A similar process is followed for the computation of the water pollution level in a catchment, as an indicator of the degree of pollution based on the grey water footprint. A sub-catchment or a period within a year is characterised as an environmental hotspot when the green water scarcity, the blue water scarcity and/or the water pollution level exceeds 100 per cent.

Table 6.2.: Summary of the indices used from the Water Footprint Assessment methodology to assess the environmental sustainability of water footprints at a catchment scale. Based on Hoekstra et al. 2011.

Assessing the Environmental Sustainability of Water Footprints at a catchment scale		
Water Footprint	**Indices**	**Comments**
Green Water Footprint	**Green Water Availability (WAgreen)** $$WAgreen[x,t] = ETgreen[x,t] - ETenv[x,t] - ETunprod[x,t]$$ ETgreen: total evaporation of rainwater from land ETenv: environmental green water requirements ETunprod: evaporation in areas or periods of that year that are unsuitable for crop growth **Green Water Scarcity (WSgreen)** $$WSgreen[x,t] = \frac{\sum WFgreen[x,t]}{WAgreen[x,t]}$$ ΣWFgreen: the total of green water footprints in the catchment WAgreen: green water availability	-expressed in: [volume/time] -ETenv: green water used by natural vegetation. Assumes land used for conservation and a default value of 30% of total land use of a catchment. --- -denotes the 'fraction of appropriation' of available green water -when WSgreen=100%, then WA=0
Blue Water Footprint	**Blue Water Availability (WAblue)** $$WAblue[x,t] = Rnat[x,t] - EFR[x,t]$$ Rnat: natural run-off in the catchment EFR: environmental flow requirements **Blue Water Scarcity (WSblue)** $$WSblue[x,t] = \frac{\sum WFblue[x,t]}{WAblue[x,t]}$$ ΣWFblue: the total of blue water footprints in the catchment WAgreen: blue water availability	-expressed in: [volume/time] -Rnat=Ract + WFblue(total) -if WFblue>WAblue, then EFR<0 --- -time-dependant; varies through the year -monthly based calculations sufficient to show variations -should be additionally assessed against the water stocks (e.g. groundwater)
Grey Water Footprint	**Water Pollution Level (WPL)** $$WPL[x,t] = \frac{\sum WFgrey[x,t]}{Ract[x,t]}$$ ΣWFgrey: the total of grey water footprints in the catchment Ract: actual run-off from the catchment	-measures the degree of pollution -if WPL=100%, assimilation capacity=0 -time-dependant; varies through the year -monthly-based calculations sufficient to show variations

Following on the identification of hotspots, the sustainability of a process is assessed against two criteria: (i) geographical context and (ii) characteristics of the process. For the former, a process is not sustainable when is situated in a spatial or temporal hotspot. For the latter, a process is unsustainable when its water footprint can be reduced or avoided altogether. Nonetheless, no criteria exist for the assessment of the sustainability of single processes. For catchment-scale studies, the assessment of the sustainability of a process is relative and dependant on the local conditions.

For assessing local impacts relating to water footprints and water scarcity, a number of Water Footprint Impact Indices have also been developed. The green, the blue and the grey water footprint impact indices ($WFIIgreen$, $WFIIblue$ and $WFIIgrey$) are aggregated and weighted measures of the environmental impact of the green, blue and grey water footprints respectively. They can be computed according to the following equation:

$$WFII_i = \sum_t \sum_t (WF_i[x,t] \times WS_i[x,t])$$

where:

i refers to the green, blue or grey component of the water footprint methodology

$WF_i[x,t]$ is the water footprint of a product/process specified by a catchment x and by month t

$WS_i[x,t]$ is the water scarcity by catchment and by month.

As noted in the Water Footprint Assessment Manual (Hoekstra et al. 2011), these indices only give a crude impression of the local environmental impacts of water footprint as a whole, which can be useful for comparative studies among catchments or as impact indices for LCA studies. For the formulation of catchment-specific strategies and the assessment of sustainable water use, the application of the volumetric accounts of the WFA methodology are suggested.

The concept of Environmental Flow Requirements (EFR) is at the centre of the WFA methodology for identifying hotspots and assessing the impacts of blue water consumption. According to the Brisbane Declaration (2007), EFR is defined as 'the quantity, timing and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihood and well-being that depend on these ecosystems'. There more than 200 methods used worldwide to calculate the EFR needed to maintain healthy riverine ecosystems (Tharme 2003), which can be grouped in four categories: hydrological approach, hydraulic rating, habitat simulation and holistic methods. The increasing interest of the hydrological community in a method for explicit consideration of EFR in hydrological assessments has driven a lively discussion in literature (Liu et al. 2016; Zeng et al. 2013; Hoekstra et al. 2012; Poff et al. 2010; Smakhtin et al. 2004). Hoekstra et al. (2012) assumed EFR to be 80% of the total water resources for the assessment of the global water scarcity; a simplistic assumption which did not consider the complexity of river regimes at a regional scale (Liu et al. 2016).

The concept of Water Footprint and the indexes introduced by the WFA methodology have been recently used as the basis for the creation of a water scarcity indicator which simultaneously considers EFR, Water Quantity and Water Quality. The quantity, quality and EFR indicator (QQE indicator) is introduced as a holistic and rapid method for assessing water stress at a regional scale (Liu et al. 2016). The following equations are used to construct the QQE water scarcity indicator:

$$S_{qqe} = S_{quantity}\,(P)/S_{quality}$$
$$S_{quantity} = BWF/BWA = (W\ x\ R)/\ (BWR - EFR)$$
$$S_{quality} = GWF/BWR$$

where:

S_{qqe} is the overall water scarcity index
$S_{quantity}$ is the index of water quantity scarcity
$S_{quality}$ is an index that quantifies the pollution-based water scarcity
P is the percentage of EFR in total blue water resources BWR to maintain "good" habitat quality
BWF is the blue water footprint (m^3)
BWA is the blue water availability (WA_{blue}) (m^3)
W is the blue water withdrawal
R is the water consumption ratio
GWF is the grey water footprint (m^3).

The discussion on the use of the QQE indicator as part of the water inventory of the Catchment PIOT and the performance of its computations for the Poole Harbour catchment to follow (section 6.5.).

6.2. System Dynamics

Systems Dynamics (SD) is an approach for understanding the dynamic behaviour of systems (Williams and Hummelbrunner 2011). It was originally introduced in 1950s by Jay Forrester and, despite its grounds on engineering and management, it was intended for the analysis of social systems. SD is based on the idea that systems consist of elements that, at a specific point in time, have a value ('stock') which can change over time through inflows and outflows. The dynamic behaviour of a given system can be explored and explained by the relationships between the stock and flows variables. Their applications to date show that SD models offer valuable insights into the dynamic behaviour of complex systems, mainly because they can provide evidence on what actually produces their behaviour (Williams and Hummelbrunner 2011).

The building blocks of an SD model are the Causal Loop Diagrams (CLDs) which serve as a language for articulating the dynamic, interconnected relations developed in a complex system. CLDs are based on the concept of 'feedback loops', which can be described as a closed sequel of causes and effects (Flood 1999). There are two types of feedback loops: positive or reinforcing (i.e. all variables respond in the same direction) and negative or balancing (i.e. at least one of system's variables in the opposite direction). The combination of multiple feedback loops for a given system results in a causal network. This network leads to the creation of the SD diagram or model; thus, enables the analysis of the interaction of the multiple variables of a system.

Recent works show a growing interest in the application of System Dynamics (SD) in water research. A number of articles have been published only in the last few years (Sanga and Mungatana 2016; Balali and Viagii 2015; Elshafei et al. 2015; Niazi et al. 2014), providing analyses on the emerging modelling approach along with SD models aimed for effective and sustainable water resources management. The SD models created intend to identify the relationships between the components of complex water systems, either at a catchment or aquifer level. The works show evidence that SD is a promising tool for exploring alternatives for effective water resources management, as it allows for explaining complex relationships among different variables of a given water system. It is a modelling approach which not only allows the investigation of natural/ecological systems, such as the interaction between surface and groundwater, but also enables the thorough analysis of the synergies developed in 'coupled' systems. The coupling of ecological and economic modelling has been greatly benefited by the SD framework, as it proves to be an effective tool for defining the interconnections and complementarities among hydrological, economical and sociological variables. This holistic view on a water system enables the investigation of trade-offs relevant to decision-making and the selection of the instruments allowing the design of truly sustainable, optimum solutions. Nevertheless, there are several limitations which need to be addressed in future applications of the SD framework in water research (Sanga and Mungatana 2016): seasonal variations of climatic parameters need to be included in the modelling; the frameworks and models produced should be grounded on robust assumptions regarding the social aspects of the systems under study; the institutional structures and relevant policies should be modelled as part of the system.

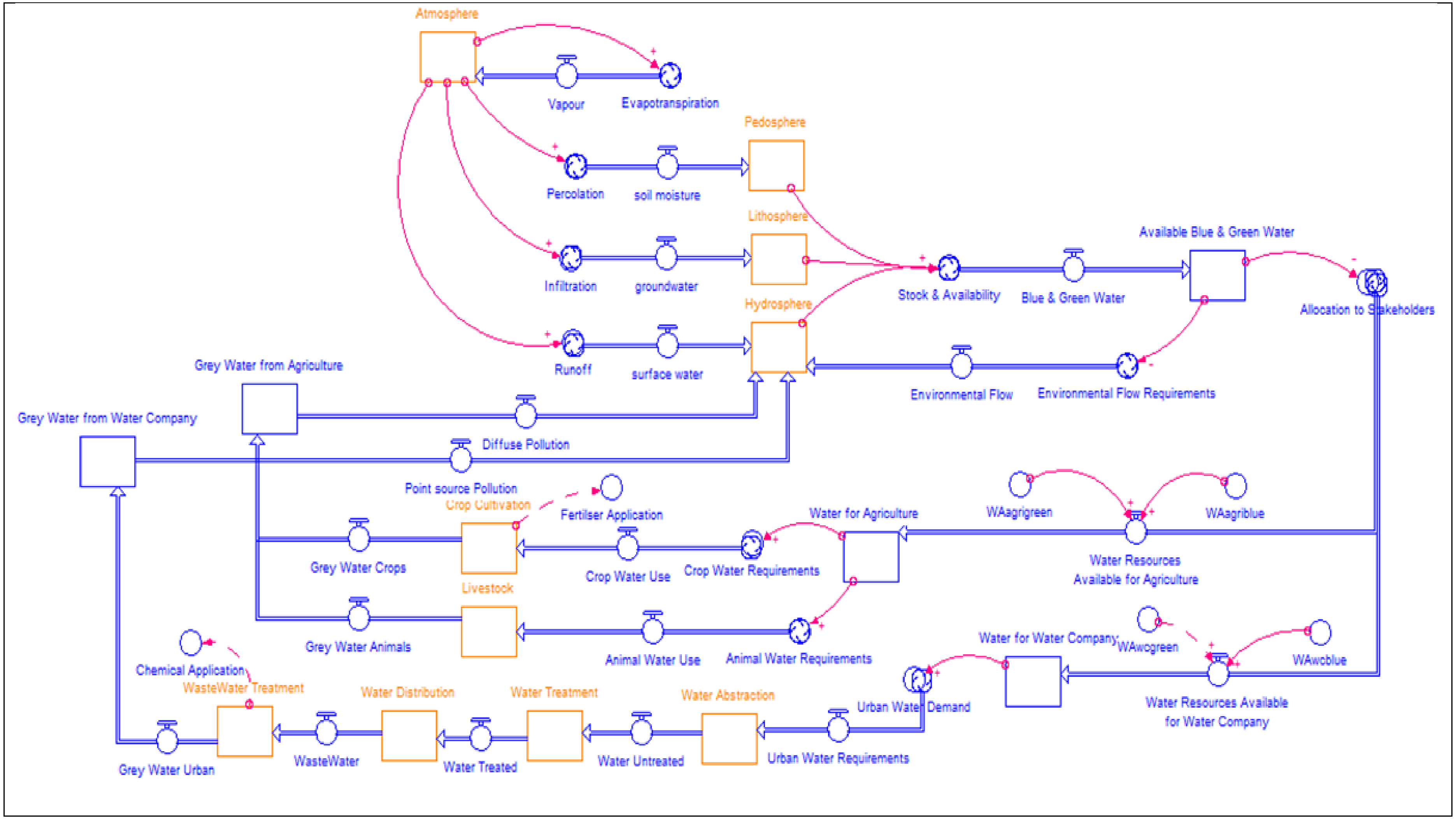
The SD framework has been employed for the identification of the causal relationships among the actors of the catchment. The SD-based representation of the catchment is shown in **Figure**

6.9.(a,b). The SD diagram has been developed using STELLA 10.1.2., which is a software designed by a US-based company (isee systems) for modelling the dynamics of highly interdependent systems. The STELLA software provides a set of simple building blocks that enable the representation of multiple systems, with a range of applications in environmental sciences.

Causal loop diagrams represent each of the actors, from a 'sectorial' perspective, which enables the translation of the Catchment Physical Input-Output Table into a systems dynamics representation. Each of the catchment's sectors are depicted as a 'stock'. Then, a main causal loop diagram is created, which reveals the relationship among actors. That is, the processes underpinning the water cycles occurring within the catchment boundaries are depicted, along with the resulting type of water and the demand it fulfils. For example, the 'Available Blue and Green Water' satisfies the need of the actor Ecosystem/ sector Hydrosphere for Environmental Flow Requirements (EFR), which provides the minimum environmental for maintaining an optimum status within the Hydrosphere.

The SD model produced is a simplified representation of the internal structure of the complex catchment system in terms of the interconnections among its actors and their sectors. As such, it is a pictorial representation of the systems' behaviour, which will be further studied for a number of different scenarios (section 6.4.1.). It illustrates the interactions among the multiple water cycles occurring within the catchment boundaries; thus, the interactions between the ecosystem, the sector of agriculture and selected built assets. The identification of the interdependent relations drives the selection of the indices which best describe the flows of the catchment and, predominantly, the outputs of the stakeholders or of their sectors to the other subsystems of the catchment. For example, the metric of Grey Water Footprint or the index of Water Pollution Level are identified as the tools applicable for the quantification of the outputs of the sectors Crop Cultivation and Livestock to the sector of Hydrosphere.

The joint use of the SD catchment model with water accounting and impact assessment methods would provide evidence on the overall performance of the catchment system under the implementation of different strategies. This would be particularly interesting for the strategies involving a combination of land management approaches and conventional end-of-pipe solutions to tackle environmental issues. It would be an effective methodology to identify synergies and trade-offs at a system's level and quantify outputs for multiple actors. Its transparent structure and flexible modelling rules enable the study of the catchment as a whole system, the identification of hotspots for further study or the further expansion of the model. Further, the SD model created can serve as the basis for embedding mathematical formulas and programme coding, in order to automatically compute the outputs among stakeholders at a catchment scale, for multiple time scales.


Atmosphere
Vapour
Evapotranspiration
Pedosphere
Percolation
soil moisture
Lithosphere
Infiltration
groundwater
Hydrosphere
Runoff
surface water
Stock & Availability
Blue & Green Water
Available Blue & Green Water
Allocation to Stakeholders
Environmental Flow
Environmental Flow Requirements
Grey Water from Agriculture
Grey Water from Water Company
Diffuse Pollution
Point source Pollution
Crop Cultivation
Fertilser Application
Grey Water Crops
Crop Water Use
Crop Water Requirements
Water for Agriculture
WAagrigreen
WAagriblue
Water Resources
Available for Agriculture
Livestock
Grey Water Animals
Animal Water Use
Animal Water Requirements
Water for Water Company
WAwcgreen
WAwcblue
Water Resources Available
for Water Company
Urban Water Demand
Chemical Application
WasteWater Treatment
Water Distribution
Water Treatment
Water Abstraction
Grey Water Urban
WasteWater
Water Treated
Water Untreated
Urban Water Requirements

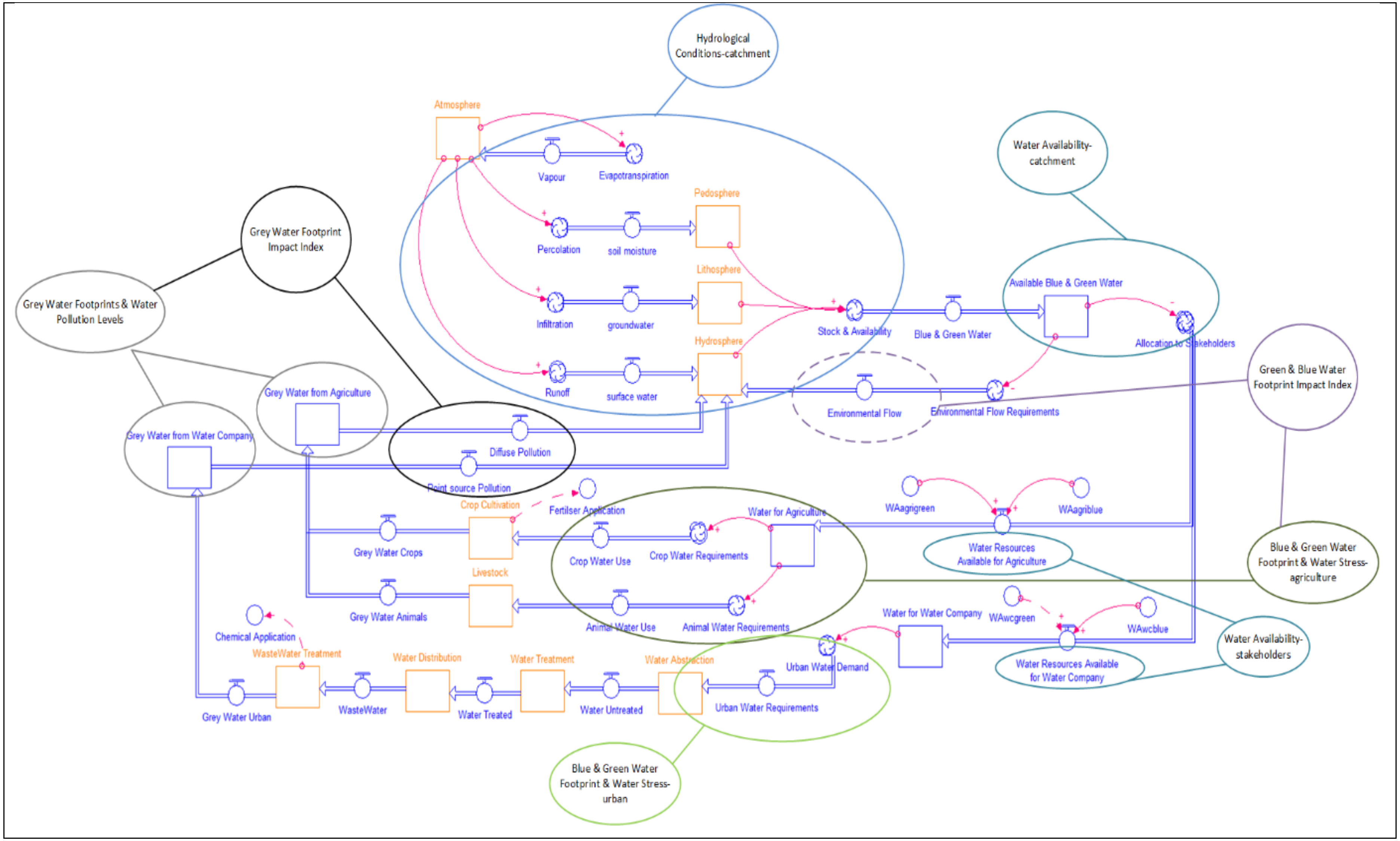


Figure 6.9.: (a) The System Dynamics model for the Poole Harbour Catchment. The model serves methodological purposes: it shows which relationships among the stakeholders of the catchment need to be quantified and thus, drives the selection of relevant water indices, as presented in figure section (b).

6.3. Water Accounts for the Catchment's Stakeholders

This section presents and analyses the methodology formulated and applied for the computation of the water regimes and the process outputs for each of the catchment's stakeholders (or actors). The methodologies created for the computations is based on literature and is reproducible for multiple catchments and case studies. Herein, the application of the methodologies is demonstrated for the selected case study, the Poole Harbour Catchment. Details on the data requirements and data sources per actor are also provided, whilst the assumptions made are clearly mentioned.

For the Poole Harbour Catchment, the stakeholder analysis has identified three main actors: the ecosystem, the water company and the agricultural sector. For each of the actors, an approach for the construction of their water inventory is firstly presented, followed by the methodology for the computation of environmental outputs. The latter is based on the systems dynamic model presented earlier (section 6.2.) and the choices made are discussed separately for each of the actors. **Table 6.3.** summarises the main features of the methodologies created for the computation of the water regimes of the catchment's stakeholders, namely: method (literature) underpinning the methodology, the source and type of data applied, and assumptions made.

The design of the Catchment PIOT as an output table, suggests that, only outputs (i.e. figures underpinning the computation of water-related impacts at a later stage) will be displayed on its final format. Nevertheless, the inventory of each of the actors needs to be presented as it constitutes the underpinning work leading to the computation of the outputs.

Table 6.3.: Summary of the methodologies created for performing Water Accounting for the main actors of the Poole Harbour Catchment: ecosystem, water company, agriculture. The underpinning methods and the relevant literature, the data sources and types and the assumptions made are listed. The methodologies created can be generalised and reproduced for multiple case studies/catchment typologies. The assumptions were made due to lack of catchment-specific data and limited time of the research project.

Water Accounting for the Poole Harbour Catchment: methods, data, assumptions				
Actor	**Method & Literature**	**Data Source**	**Data Types**	**Assumptions**
Ecosystem	Water Budget (McMahon et al. 2013; Fandel 2012) Montana Method (Tennant 1976) as described/modified in Liu et al. 2016 and Arthington 2012	Meteorological Office Rainfall and Evaporation Calculation Scheme (MORECS) National River Flow Archive (NRFA)	Time period: 1996-2015 MORECS weekly values for squares 180 & 181, and following hydrometric parameters: Rainfall, potential and actual evaporation, effective precipitation, soil moisture deficit Mean Annual Flow data for rivers Frome and Piddle	Annual Catchment Average computed as [52%*(sq180 values +48%* (sq181 values)] ΔS (change in soil moisture)=0 on an annual basis Catchment Mean Annual Runoff computed as the sum of Mean Annual Runoff of rivers Frome and Piddle
Water Company	Water Footprint Assessment (Hoekstra et al. 2011) modified according to Morera et al. 2016 and Manzardo et al. 2016 Life Cycle Assessment (LCA)	Wessex Water Services Ltd Literature (Morera et al. 2016)	Time period: 2010-2015 Daily & monthly values for: water quality, chemical usage, electricity consumption	All Water Treatment Plants (WTPs) and Wastewater Treatment Plants (WWTPs) of the catchment include a nitrogen-removal process and operate in the same efficiency No evaporation during treatment processes Cmax N=50 mg/l Cnat N=0 mg/l Only WTP and WWTP are accounted as part of the WF of the urban water cycle- distribution is excluded from the accounting
Agriculture	Water Footprint Assessment (Hoekstra et al. 2011) CROPWAT (FAO 2012)	National Statistics (www.ons.gov.uk) Nix (2015) CLIMWAT (FAO 2012) Nitrogen Reduction Strategy (Environment Agency 2013) Webpage: criddles.co.uk	Average UK Annual Yield data Average Fertiliser application per crop type Crop rotation for the UK	Agricultural practice and yield as in literature/across UK Bournemouth & Exeter CLIMWAT data represent the data from MORECS squares 180 & 181 respectively WF only during the production phase – no WF of fertilisers etc.

6.3.1. Actor: Ecosystem

The natural water cycle represents the water regime of the actor "ecosystem". The computation of the natural water balance is performed at a catchment scale, where the water budget is assumed to be balanced over a long period of time (Fandel 2012). In order to study the catchment as a hydrological system, the volume of the water circulating among the different natural reservoirs needs to be quantified. The basic mathematical equation used for the computation of the water budget for a hydrological catchment system is described below (McMahon et al. 2013; Fandel 2012):

$$P = ET + I + R + \Delta S \qquad (1)$$

where:

P is the mean annual precipitation in the catchment

ET is the mean annual evapotranspiration of the catchment

I refers to water stored in aquifers as a result of infiltration

R is the mean annual runoff

ΔS is the change in soil moisture storage over the analysis period

Values are given in millimetres of rainfall (mm) or cubic meters of water (m^3). The budget is computed over a period of a year (12 months). At this annual time step, the change in soil moisture (ΔS) is assumed zero (Wilson 1990). Evapotranspiration (ET) is defined as the sum of evaporation (i.e. liquid water transferred as water vapour to the atmosphere) and transpiration (i.e. evaporation from within the leaves of a plant) from soil surfaces (Allan 2003). Literature distinguishes two types of evapotranspiration for water studies: actual ($ET actual$) and potential ($ET potential$). The latter can be defined as the "rate at which evapotranspiration would occur from a large area completely and uniformly covered with growing vegetation which has access to an unlimited supply of soil water, and without advection or heating effects" (Dingman 1992), whilst actual evapotranspiration refers to the quantity of water that is transferred to the atmosphere from an evaporating surface (Wiesner 1970). As recently discussed in literature (McMahon et al. 2013), a plethora of theoretical and experimental methods exist for the estimation of both types of evapotranspiration; however, their use is still largely debated. The term 'Effective Precipitation' (EP) refers to the amount of water that is actually available to feed runoff (R) and infiltration (I) (Fandel 2012). Thus, equation (1) can also be written as:

$$P = ET + EP + \Delta S \qquad (2)$$

where:

$$EP = I + R \qquad (3).$$

For the computation of the natural water budget of the Poole Harbour Catchment, hydrological data were retrieved from the Meteorological Office Rainfall and Evaporation Calculation Scheme (MORECS) and the National River Flow Archive (NRFA). The data were processed as analysed below with the end goal to compute annual and seasonal values at a catchment scale.

The MORECS scheme was introduced in late 1970s (Thompson et al 1981) for providing real-time soil moisture deficit data and as a replacement of the previous system, the Estimated Soil Moisture Deficit (ESMD) bulletin. The revised and refined version of MORECS (version 2.0) is currently used to produce hydrological outputs on a grid square of 40 x 40 Km (**Figure 6.10.a**). Briefly, MORECS uses daily synoptic weather data to provide grid square average estimates of weekly and monthly hydrological values (actual evaporation, hydrologically effective rainfall and soil moisture deficit) under the British climatic conditions, using a comprehensive approach (Hough and Jones 1997). The meteorological data inputs (sunshine, temperature, vapour pressure, wind speed and rainfall) are obtained from 125 synoptic stations distributed across the UK (**Figure 6.10.b**). Values are estimated for each grid-square, based on the objective interpolation. Input data are derived from the nearest, to the grid, 3 to 6 stations. Inverse distance weighting is used when there are less than 3 stations. Although the MORECS scheme originally assumed a single type of land use (grassland), the revised version assumes a diversity of surfaces, such as urban and forestry areas and different crops. Further details concerning the mathematical structure of the MORECS scheme, the processes followed to normalise or standardise station data and the assumptions regarding the computation of the values of the water balance of each grid square can be retrieved in literature (Smith et al. 2006; Hough and Jones 1997).

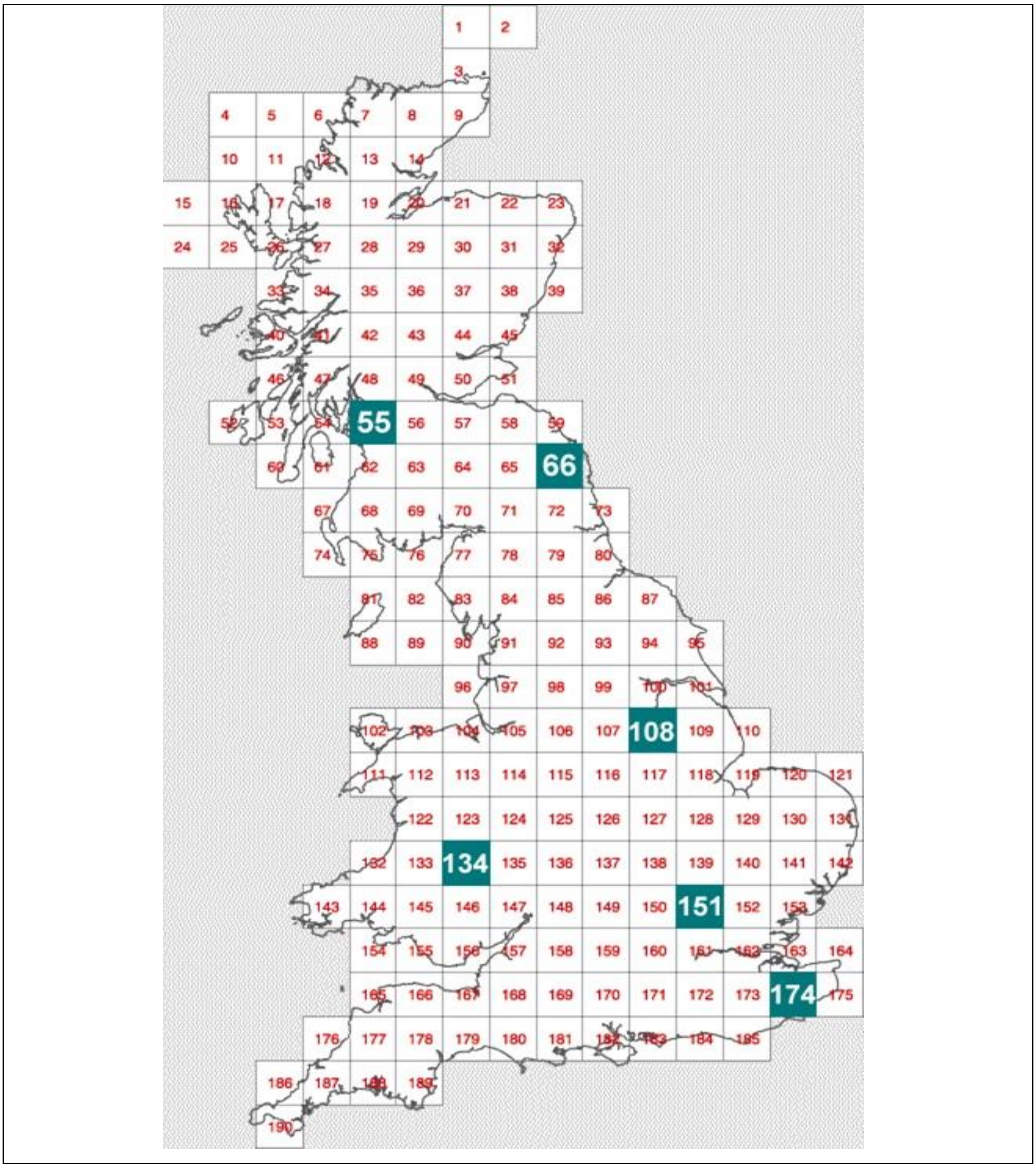
In the MORECS grid matrix, the Poole Harbour Catchment is positioned between two grid squares, namely squares 180 and 181 (**Figure 6.11**.). As shown in the map, 52% of the catchment's surface is within the square number 180 and the rest 48% in the square number 181. For this spatial pattern, a mathematical formula was introduced for the computation of the MORECS hydrometric parameters (rainfall, actual evapotranspiration, potential evapotranspiration, effective rainfall) of the Poole Harbour Catchment. At the catchment scale considered, the hydrometric parameters (HP) can be derived as follows.:

$$HP@catchment\ scale = \{(0.52 * [sq180values]) + (0.48 * [sq181values])\} \quad (4)$$

where:

$HP@catchment\ scale$: hydrometric parameter computed at the catchment scale

$[sq180values]$, $[sq181values]$: processed data for MORECS squares 180 and 181 respectively.


1 2
3
4 5 6 7 8 9
10 11 12 13 14
15 16 17 18 19 20 21 22 23
24 25 26 27 28 29 30 31 32
33 34 35 36 37 38 39
40 41 42 43 44 45
46 47 48 49 50 51
52 53 54 55 56 57 58 59
60 61 62 63 64 65 66
67 68 69 70 71 72 73
74 75 76 77 78 79 80
81 82 83 84 85 86 87
88 89 90 91 92 93 94 95
96 97 98 99 100 101
102 103 104 105 106 107 108 109 110
111 112 113 114 115 116 117 118 119 120 121
122 123 124 125 126 127 128 129 130 131
132 133 134 135 136 137 138 139 140 141 142
143 144 145 146 147 148 149 150 151 152 153
154 155 156 157 158 159 160 161 162 163 164
165 166 167 168 169 170 171 172 173 174 175
176 177 178 179 180 181 182 183 184 185
186 187 188 189
190

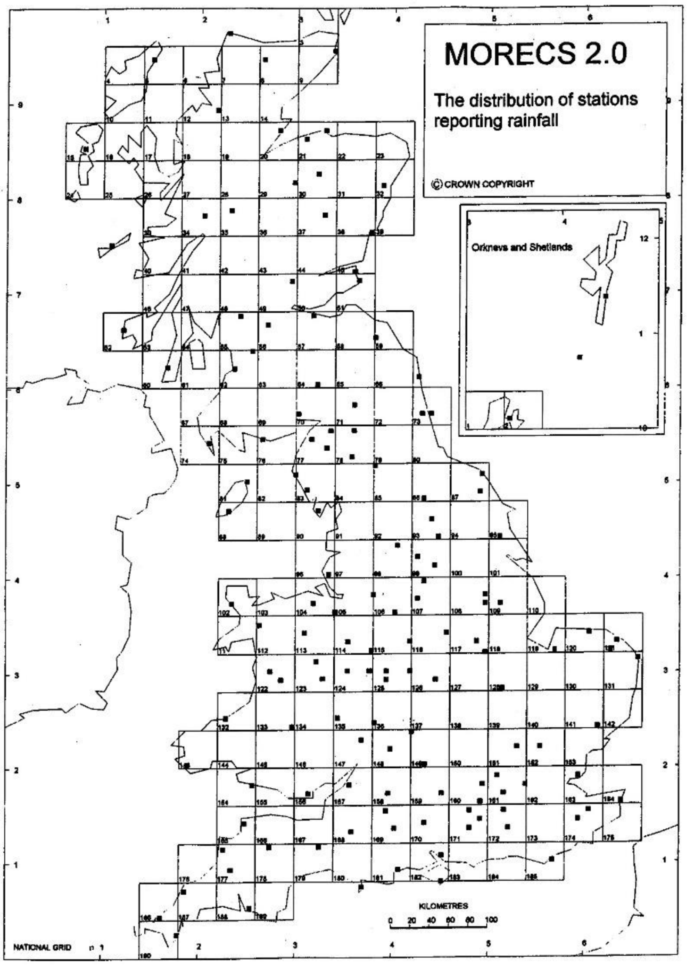


Figure 6.10.: (a) MORECS squares in the United Kingdom, (b) the distribution of the meteorological stations within the MORECS grid.

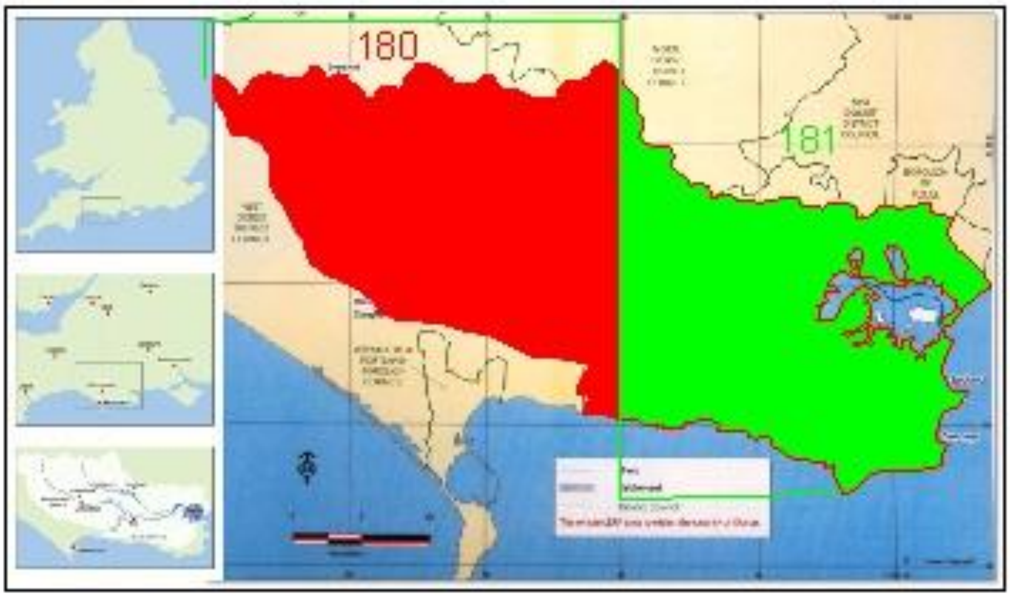


Figure 6.11: Poole Harbour Catchment boundaries as distributed in the MORECS grid squares. 52% of the catchment's surface is within square 180 and 48% within square 181.

MORECS data for the grid squares 180 and 181 were provided by the Met Office, under the license agreement they share with the project's industrial partner. The data were provided for the period 1996-2015 in weekly values for the following hydrometric parameters: rainfall, potential evaporation, actual evaporation, effective precipitation and soil moisture deficit.

The MORECS data were processed for each square and the formula (4) was used for the computation of catchment-based values. The following procedure was followed: the sum of weekly values produced the annual value for each square (Annual SUM). Then, the average of the annual values was computed (Square Average). The Square Average values were then used for the computation of the catchment values, which represent the mean annual average (Annual Catchment Average).

The same methodology was applied to all hydrometric parameters of the catchment, except for the soil moisture deficit (SMD). This is described by the change of the soil moisture and is assumed zero ($\Delta S = SMD = 0$) for a hydrological year of 12 months, as described in literature (McMahon et al. 2013). The primary data and the computations performed are presented in the Appendix (APPENDIX A). The Annual Catchment Average data are presented below (**Table 6.4.**) in millimetres (mm) of rainfall. Due to the seasonal character of the hydrometric parameters and their influence on agricultural activities, the seasonal averages were also computed. The hydrological year was divided in a dry (April-September) and a wet (October-March) period. The computational methodology followed was identical to that formulated for the annual averages. **Table 6.4.** shows the seasonal hydrometric values of the Poole Harbour Catchment for the dry and the wet seasons. The cross over between calendar years results in an uneven number of values between the two seasons (APPENDIX A).

Table 6.4.: Annual and Seasonal Catchment Average hydrometric parameters for the Poole Harbour Catchment. The computations are based on data retrieved from the MORECS scheme, for the period 1996-2015.

Hydrometric Parameter	**Annual Catchment Average** Period: 1996-2015	**Seasonal Catchment Averages** Period: 1996-2015	
	(mm)	**Dry Season: April-September (in mm)**	**Wet Season: October-March (in mm)**
Rainfall ($\boldsymbol{P}$)	881.6	372.2	550.5
Potential Evapotranspiration ($\boldsymbol{ETpotential}$)	545.7	424.0	145.7
Actual Evapotranspiration ($\boldsymbol{ETactual}$)	501.9	302.0	141.0
Effective Precipitation ($\boldsymbol{EP}$)	379.7	70.2	409.5

Table 6.5.: Annual and Seasonal mean flow and runoff for the Poole Harbour Catchment. The computations are based on data retrieved from the National Flow River Archive (NFRA), for the period 1996-2015.

Mean Flow and Runoff values (period: 1996-2015)						
	Annual		**Dry Season (April-September)**		**Wet Season (October-March)**	
	$Qannual$ (m^3/s)	R (* 10^6 m^3)	$Qseason$(m^3/s)	R(* 10^6 m^3)	$Qseason$(m^3/s)	R(* 10^6 m^3)
River Frome	6.671	210.4	4.7	147.1	8.8	278.3
River Piddle	2.474	78	1.7	54.5	3.6	112.9
Poole Harbour Catchment: $\boldsymbol{Rcatch}$	**288.4**		201.9		391.4	

For a more elaborate hydrological water budget of the Poole Harbour Catchment, according to equations (1) and (2), the hydrometric parameter of Effective Precipitation (EP) was further analysed according to equation (3). The runoff (R) was computed based on the runoff of the two major rivers occurring in the catchment: the Frome and the Piddle. Data for the mean annual flow $(Qannual)$ of the two rivers were retrieved from the National River Flow Archive (NRFA), for the period 1996-2015. These data are freely available for research projects. The datasets provided are based on gauged data, measured in cubic meters per second (m^3/s). The conversion of the flow data into volume of water (runoff), was based on the following mathematical formula:

$$R\ = (Qannual) * 3600 * 24 * 365 \qquad (5).$$

The sum of the two values produced was assumed to describe the mean annual runoff of the Poole Harbour Catchment (**Table 6.5.**). Then, the catchment's mean annual infiltration was computed applying equation (3). Same methodology was applied for the computation of the mean seasonal flow $(Qseason)$ and seasonal runoff of the catchment for the dry (April to September) and the wet (October to March) periods.

The complete hydrological budget of the Poole Harbour Catchment at both annual and seasonal basis is presented in **Table 6.6.**, described in millimetres (mm) of rainfall, volume of water (m^3) and ratio (%) of each of the hydrometric parameters compared to annual catchment precipitation. The volume of water (m^3) is computed when multiplying the values in (mm) of rainfall with the size of the catchment ($820Km^2$). Only the actual evaporation values are used to described the water budget.

From the total rainfall (881.6 mm) entering the catchment, 57% evaporates. This value is slightly lower than the European average actual evapotranspiration rates that is equal to 60% of the total rainfall (McMohan et al. 2013). The rest 43% represents the mean annual effective precipitation, which equals to 311.4 million cubic meters ($311.4 * 10^6$ m^3) of water. The mean annual runoff and mean annual infiltration represent nearly 93% and 7% of the mean annual effective rainfall respectively.

The hydrometric parameters defer significantly at a seasonal level. For example, the actual evapotranspiration rate differs by nearly 55% between the values estimated for the dry (April-September) and the wet (October-March) periods in the Poole Harbour Catchment. Further discussion on the seasonal variability of the catchment's hydrology to follow (sub-section 6.3.3.).

Table 6.6.: The Natural Water Balance for the Poole Harbour Catchment. The computations are based on data retrieved from the MORECS scheme (Met Office) and the National Flow River Archive (NFRA), for the period 1996-2015. The values are presented in millimetres of rainfall (mm), volume of water (m^3) and ratio (%) to the total precipitation.

Natural Water Balance: Poole Harbour Catchment (period: 1996-2015)								
		P	=	***ET***	+	***I***	+	***R***
Annual	Rainfall (mm)	881.6	=	501.9	+	26.6	+	353.1
	Volume ($*10^6$ m^3)	723	=	411.0	+	23.6	+	288.0
	Ratio (%)	100	=	57.0	+	3.2	+	39.8
Dry Season (April-September)	Rainfall (mm)	372.2	=	302.0	+	0.0	+	70.2
	Volume ($*10^6$ m^3)	305.2	=	247.6	+	0.0	+	57.6
	Ratio (%)	100	=	81.1	+	0.0	+	18.9
Wet Season (October-March)	Rainfall (mm)	550.5	=	141.0	+	53.7	+	355.3
	Volume ($*10^6$ m^3)	451.0	=	115.6	+	44.0	+	391.4
	Ratio (%)	100	=	25.6	+	9.8	+	86.7

The Environmental Flow Requirements (EFR) for the two main rivers of the Poole Harbour Catchment have been computed based on the Tennant (1976) or "Montana method", as described by more recent literature (Liu et al. 2016; Arthington 2012). This method solely relies on the recorded or estimated flow regimes for the calculation of EFR. Thus, it is a rather rapid and simple method which has been applied in diverse case studies, for different parts of the world (e.g. Pastor et al. 2014; Men et al. 2014; Arthington 2012; Kumara and Srikantaswamy 2011).

The mathematical equations to describe the Montana method are as follows:

$$EFR = \sum_{i=1}^{12} e_{ij} \qquad (6)$$

$$e_{ij} = 3600 * 24 * n_i * Q_i * P_{ij} \qquad (7)$$

where

e_{ij} is EFR in month i at habitat quality level j

n_i is the number of days in month i

Q_i (m^3/s) is the mean daily flow in month i

P_{ij} (%) is the percentage of the mean annual flow in month i at habitat quality level j.

The term EFR_j (m^3) is the annual environmental flow requirement at a river fish quality level j. Literature (Arthington 2012; Tennant 1976) suggests a set of values (**Table 6.7.**) to describe the temporal variation in the temporal proportion of the mean annual flow to maintain different levels of fish-habitat conditions.

Daily flow data over the period 1996-2015 were retrieved from the National River Flow Archive (NRFA) for the rivers Frome and Piddle. It was assumed that the sum of the EFR requirements of these two aquatic systems would represent the EFR of the Poole Harbour Catchment as a whole (**Table 6.8.**). For each of the rivers, the mean monthly flow ($Qmonthly$)(m^3/s) was estimated based on the gauged daily flow values. Then, the monthly EFR values were computed for two different habitat quality levels j: optimum ($P_{ij} = 0.8$) and good ($P_{ij} = 0.2$ for the wet period and $P_{ij} = 0.4$ for the dry period). The annual EFR equals to the sum of the monthly EFR values. The average of the annual values for the period 1996-2015 describes the EFR of each of the rivers in the Poole Harbour Catchment. The mean monthly flow and EFR values computed for the Frome and Piddle rivers can be accessed in the Appendix (APPENDIX A).

Table 6.7.: Temporal variation in the range of proportions of the mean flow that must be allocated to maintain various levels of habitat quality. Adapted from Liu et al. 2016. Based on Tennant (1976) and Arthington (2012).

Flow category or habitat quality	Recommended Flow (% of mean flow)	
	October to March	April to September
Maximum	200	200
Optimum	60-100	60-100
Outstanding	40	60
Excellent	30	50
Good	20	40
Moderately degraded	10	30
Highly to severely degraded	≤10	≤10

Table 6.8.: Annual Environmental Flow Requirements (EFR, m^3) of the Poole Harbour Catchment. The calculations are based on the Montana method and the data have been retrieved from the National River Flow Archive, for the period 1996-2015. It is assumed that the sum of the EFR of the two main river systems represent the catchment's annual EFR. The EFR were estimated for two habitat quality levels j: Optimum and Good.

Environmental Flow Requirements (EFR) Period: 1996-2015		
	EFR (m^3)	
	j= Good	*j*= Optimum
River Frome	$57 * 10^6$	$17 * 10^7$
River Piddle	$1.8 * 10^6$	$6.4 * 10^7$
Poole Harbour Catchment	$\mathbf{58.8 * 10^6}$	$\mathbf{23.4 * 10^7}$

The computation of the hydrological balance and of the environmental flow requirements of the Poole Harbour Catchment enable the analysis of the internal structure of the natural water environment and the processes among its “sectors”: atmosphere, pedosphere, lithosphere and hydrosphere. The results of this hydrological analysis provide the basis to compute the outputs of the actor ‘ecosystem’ to the other actors of the catchment. The Green Water Availability (WA_{green}) and the Blue Water Availability (WA_{blue}) were computed, as introduced by the Water Footprint Assessment methodology (**Table 6.1.**) The selection of these metrics is driven from a number of methodological reasons. Firstly, their structure is transparent, enabling their reproduction. They are also spatially explicit for catchment areas, while their components are relevant to hydrological figures, such as actual evapotranspiration and runoff. Further, they have been used in literature as a basis for the creation of other indicators assessing water scarcity in a holistic approach (e.g. Liu et al. 2016). The computation of the latter to follow (sub-section 6.4.).

The Green Water Availability $WA_{green}$is computed at a catchment scale according to the formula:

$$WA_{green} = ET_{green} - ET_{\mathrm{env}} - ET_{unprod} \qquad (8)$$

For the computation of the WA_{green} of the Poole Harbour Catchment at an annual rate, the following assumptions were made:

$$ET_{green} = ET_{actual}$$

$$ET_{env} = 30\% * ET_{actual}$$

$$ET_{unprod} = ET_{potential} - ET_{actual} \,.$$

Thus, (8) is re-written as:

$$WA_{green} = ET_{actual} - (0.3 * ET_{actual}) - (ET_{potential} - ET_{actual}) \quad (9)\,.$$

The annual $WA_{green}$of the Poole Harbour Catchment is computed based on the hydrological figures provided (**Table 6.2.**; **Table 6.4.**; **Table 6.5.**) equal to $\boldsymbol{WA_{green} = 251.5 * 10^6\, m^3/year}$.

The Blue Water Availability $WA_{blue}$is computed at a catchment scale, for habitat quality level j, according to the formula:

$$WA_{blue} = R_{nat} - EFR_j \qquad (10).$$

For the Poole Harbour Catchment, (10) needs to be amended to include the water volume transferred from and to other catchments (TR_{in} and TR_{out} respectively), as depicted in the water resources analysis map (**Figure 6.9.**). Thus, (10) is re-written as:

$$WA_{blue} = R_{nat} - EFR_j + TR_{in} - TR_{out} \quad (11).$$

According to the figures provided from the industrial partner, the annual volume of water transferred to and from the catchment is equal to $TR_{in} = 1.6 * 10^6\, m^3$ and $TR_{out} = 5.2 *$

$10^6\ m^3$ respectively. The computation of the annual WA_{blue} is based on the hydrological and environmental figures provided (**Table 6.5.**; **Table 6.7**) and is performed for two habitat quality levels.

The annual WA_{blue} of the Poole Harbour Catchment, to maintain the optimal habitat quality, is equal to $\boldsymbol{WA_{blue/optimal} = 47.2 * 10^6\ m^3/year.}$ The annual WA_{blue} of the Poole Harbour Catchment, to maintain good habitat quality, is equal to $\boldsymbol{WA_{blue/good} = 222.4 * 10^6\ m^3/year.}$

6.3.2. Actor: Water Company

Recent advancements and additions in the Water Footprint Assessment methodology (Hoekstra et al. 2011) for its application to the urban water cycle (Morera et al. 2016; Manzardo et al. 2016) were used for the formulation of the inventory of the actor 'Water Company'. The methodology created was aimed at the computation of the total water footprint of the urban water cycle ($WF_{urban,catch}$) occurring in the delineated area of the Poole Harbour Catchment.

A number of assumptions were made for the formulation of the methodology: the abstraction of surface or groundwater is part of the water treatment asset system; the contribution of the sector of distribution (for both water and wastewater) is considered negligible to the $WF_{urban,catch}$; the total number of Water Treatment Plants (WTPs) and Wastewater Treatment Plants (WWTPs) is equal to 18 and 23 respectively; all WTPs and WWTPs include a nitrogen removal process; the efficiency of the asset systems (WTP, WWTP) selected as examples are representative of the average efficiency of the total of asset systems located in the Poole Harbour Catchment.

The following equation presents the rationale of the methodology formulated:

$$
\begin{aligned}
WF_{urban,catch} &= WF_{WTP,catch} + WF_{WWTP,catch} \\
&= \left\{ \sum_{i=1}^{x} \sum_{j=1}^{k} WF_{WTP,total} + \sum_{i=1}^{x} \sum_{y=1}^{l} WF_{WWTP,total} \right\} \\
&= \left\{ \sum_{i=1}^{x} \sum_{j=1}^{k} [WF_{WTP,blue} + WF_{WTP,grey}] \right. \\
&\quad \left. + \sum_{i=1}^{x} \sum_{y=1}^{l} [WF_{WWTP,blue} + WF_{WWTP,grey}] \right\} [volume\ /\ time] \ (12)
\end{aligned}
$$

where:

$WF_{urban,catch}$ is the total water footprint of the urban water cycle at a catchment scale
$WF_{WTP,catch}$ is the sum of the total water footprint of all the water treatment plants of a catchment
$WF_{WWTP,catch}$ is the sum of the total water footprint of all the wastewater treatment plants of a catchment
x is the period considered for the computation, in months
k is the total number of water treatment plants located in a given catchment
l is the total number of wastewater treatment plants located in a given catchment.

For the computation of the $WF_{WTP,total}$ and its components ($WF_{WTP,blue}$, $WF_{WTP,grey}$), a WTP located within the delineated geographical boundaries of the Poole Harbour catchment, the Clarendon WTP, was selected as an example infrastructure asset system for the performance of the computations regarding the WTPs of the catchment. Its selection was driven by methodological purposes: the treatment processes of the plant include nitrogen-removal and data on water, energy and chemical use were available from the project's industrial partner for a timescale of five consecutive years (2010-1015).

The Clarendon WTP is located in Salisbury (UK) and treats 11,000 m^3 per day which is then discharged into the Cockey Down reservoir before being abstracted for distribution. The WTP is designed to serve a population of 34,000 people and contains a facility for removing organic matter and nitrogen from 'raw' (fresh) water. This is of particular importance for the Poole Harbour Catchment, where the extensive agricultural activities affect the quality of the groundwater by introducing a large number of nitrates to the aquifer systems, through the natural processes of percolation and infiltration. The Clarendon WTP operates three main chemical processes (**Figure 6.12.**): (i) Granular Activated Carbon (GAC) for the removal of chemicals relating to the odour and taste of water and of pesticides, (ii) Ion Exchange (IEX) for nitrate removal and (iii) Chlorine Disinfection (SDF) where chlorine (Cl_2) is injected in the water in order to deactivate present microorganisms. The WTP only treats groundwater abstracted from two boreholes, Borehole 1 and 2, which produce an average daily flow of 7,000 m^3 and 4,000 m^3 respectively. The data presented in this section were obtained by the industrial partner of the project (Wessex Water Services Limited) during a site visit conducted in May 2016).

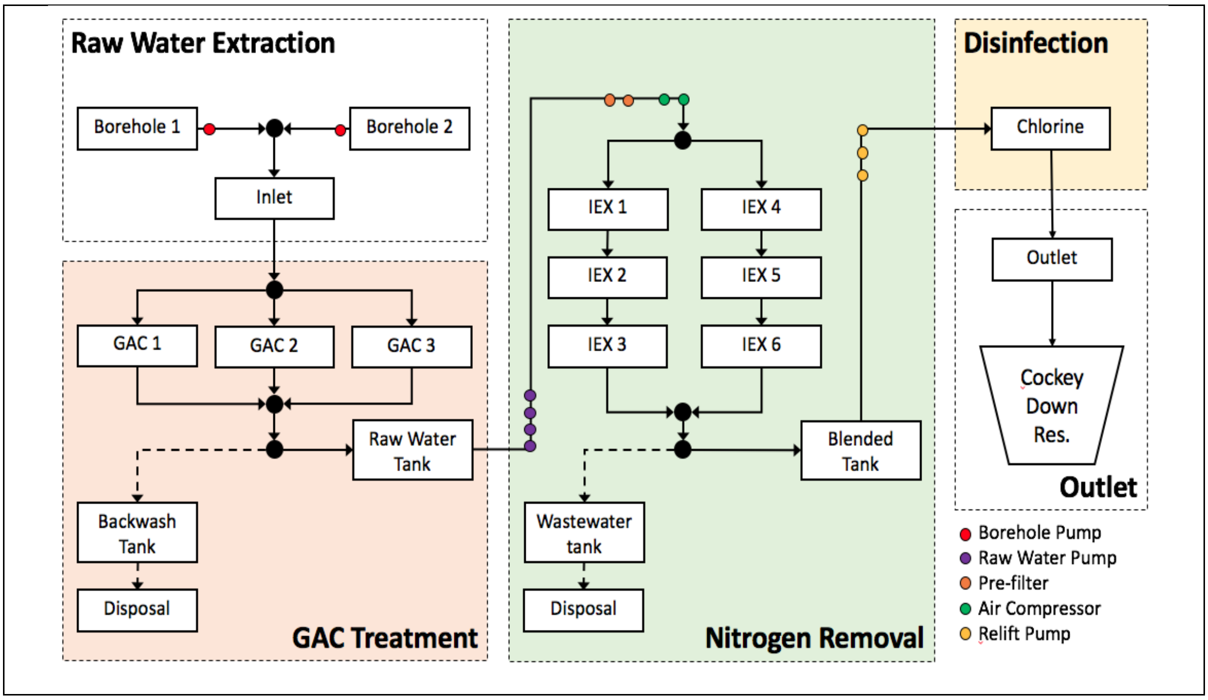


Figure 6.12.: Schematic representation of the Clarendon Water Treatment Plant, located in Salisbury, UK. The main chemical processes operating in the plant are depicted. The dotted arrows represent processes carried out periodically. Solid lines represent continuous processes. GAC stands for Granular Activated Carbon and IEX stands for Ion Exchange.

The methodology created for the computation of the $WF_{WTP,total}$ includes the computation of the blue ($WF_{WTP,blue},$) and the grey $WF_{WTP,grey}$) water footprint of the operation phase of the WTP. Further, the WTP is assumed as closed system where no water evaporation takes place. Hence, the green water footprint ($WF_{WTP,green}$) of the WTP is not computed as part of the methodology. Same approach has been followed in recent literature (Morera et al. 2016; Manzardo et al. 2016) which states that the green water footprint is limited in urban areas and the contribution of soil ('green') water to the total volume of water abstracted for the urban water cycle is negligible. Thus, the total water footprint of the WTP during its operation phase over a certain time period calculated based on the equation :

$$WF_{WTP,total} = WF_{WTP,blue} + WF_{WTP,grey} \qquad [volume/time] \quad (13).$$

The components of equation (13) are expressed as volume of water per time unit. Computations were originally performed for monthly values (30 days) and then aggregated to annual values. The latter stage is necessary for the computation of annual environmental outputs which will feed into the Catchment PIOT.

For the computation of the $WF_{WTP,blue},$ the definition of blue water footprint as described by Morera et al. (2016) is followed. This is a simplified version of the original definition of blue water footprint as defined by Hoekstra et al. (2011). Thus, $WF_{WTP,blue}$ is computed according to the equation:

$$WF_{WTP,blue} = BlueWaterIncorporation + \ LostReturnFlow \qquad [volume/time]\ (14)$$

where:

$BlueWaterIncorporation$ is the blue water incorporate in the production of chemicals ($WF_{chemicals}$) and energy (WF_{energy}):

$$BlueWaterIncorporation = \ WF_{energy} + WF_{chemicals} \quad [volume/time]\ (15)$$

$LostReturnFlow$ is the blue water not available for reuse, due to water not returning to the same catchment. It is equal to the difference between the volume of water treated (Q_{out}) and the volume of water abstracted by the groundwater resources (Q_{in}):

$$LostReturnFlow = \ Q_{out} - Q_{in} \quad [volume/time]\ (16).$$

It was assumed that there were no water loses during the water treatment process, hence: $Q_{out} = Q_{in}$ and $LostReturnFlow = 0$.

Due to the absence of literature on the computations of WFenergy and WFchemicals using the Water Footprint Assessment (WFA) methodology, these figures were computed based on the Life Cycle Assessment (LCA) approach. The LCA software SimaPro was utilised for the performance of the computations. The system boundaries were defined as the operational phase of the Clarendon WTP and the functional unit selected was that of one cubic meter of water (1m^3). **Table 6.9**. summarises the input data for the LCA computations.

The combined use of the equations proposed by Hoekstra et al. (2011) and Morera et al. (2016) served for the computation of the $WF_{WTP,grey}$. According to Hoekstra et al. (2011), the grey water footprint (WF_{grey}) is computed as follows:

$$WF_{grey} = \frac{L}{C_{max} - C_{nat}} \quad [volume/time]\ (17)$$

where:

L is the pollutant load
C_{max} is the maximum acceptable concentration for the receiving body
C_{nat} is the natural concentration of the pollutant in the receiving body.

The equation introduced by Morera et al. (2016) follows the original WF_{grey} calculation and is modified for a built asset system (infrastructure, wastewater treatment plant, WWTP). It is based on a mass balance at the discharge point. It considers that the grey water footprint is the minimum volume of water required to dilute the pollutant concentration from the WWTP effluent concentration to the maximum pollutant concentration allowed in water bodies. The mass balance of pollutants at the discharge point and the grey water footprint based on the mass balance of pollutants ($WF_{WWTP,grey}$) are calculated as follows:

$$Q_e * c_{e_{(p)}} + WF_{grey} * c_{nat_{(p)}} = \left(Q_e + WF_{grey_{(p)}}\right) * c_{max_{(p)}} \qquad (18)$$

$$WF_{grey,urban} = \max\left[WF_{grey_{(p)}}\right] = \max\left\{\frac{\left[Q_e * \left(c_{e_{(p)}} - c_{max_{(p)}}\right)\right]}{c_{max_{(p)}} - c_{nat_{(p)}}}\right\} \quad [volume/time] \ (19)$$

where:

Q_e is the effluent rate (volume/time)
$c_{e_{(p)}}$ is the concentration of a pollutant p in the effluent (mass/time)
$c_{max_{(p)}}$ is the maximum concentration of pollutant p permitted in the receiving water body
$c_{nat_{(p)}}$ is the natural concentration of a pollutant p in the receiving water body.

For the Clarendon WTP equation (19) is re-written as follows:

$$WF_{WTP,grey} = \frac{Q_{inlet} * \left(c_{inlet_{(TN)}} - c_{max_{(TN)}}\right)}{c_{max_{(TN)}} - c_{nat_{(TN)}}} \quad [volume/time] \ (20)$$

where:
Q_{inlet} is the sum of the water volume entering the Clarendon WTP from boreholes 1 and 2
$c_{inlet_{(TN)}}$ is the concentration of the inlet water in total nitrogen (TN)
$c_{max_{(TN)}}$ is the maximum concentration of total nitrogen (TN) allowed in the Cockey Down reservoir according to legislative standards
$c_{nat_{(TN)}}$ is the natural concentration of the Cockey Down reservoir in total nitrogen (TN).

The methodology formulated and presented for the computation of the $WF_{WTP,total}$ applies to other built asset systems, such as wastewater treatment plants (WWTPs) for the computation of the $WF_{WWTP,total}$. Nonetheless, for the research undertaken, computations for a selected WWTP (Poole WWTP) were not performed due to time limitations. Instead, data from literature (Morera et al. 2016) were used. It was assumed that the Poole WWTP includes the same processes - including nitrogen removal- and operates in the same efficiency as the WWTP described in the aforementioned literature. To account for the different sizes of the WWTPs, the figures provided in literature were adjusted in analogy to the population served and volume of wastewater treated. The figures produced are underpinned by a consistent methodology throughout the section relating to the urban water cycle.

Table 6.10. summarises the data used for the computations of the individual components of $WF_{urban,catch}$. The annual, total water footprint of the urban water cycle (m3/year) of the Poole Harbour Catchment is computed according to equation (13) as follows:

$$
\begin{aligned}
WF_{urban,catch} = WF_{WTP,catch} + WF_{WWTP,catch} \\
= \left\{ \sum_{i=1}^{12} \sum_{j=1}^{18} WF_{WTP,total} + \sum_{i=1}^{12} \sum_{y=1}^{23} WF_{WWTP,total} \right\} \\
= \left\{ \sum_{i=1}^{12} \sum_{j=1}^{23} \left[WF_{WTP,blue} + WF_{WTP,grey} \right] \right. \\
\left. + \sum_{i=1}^{12} \sum_{y=1}^{23} \left[WF_{WWTP,blue} + WF_{WWTP,grey} \right] \right\} = 1.55 * 10^{8}\ m^{3}/year.
\end{aligned}
$$

The outputs produced for the Catchment PIOT assume total outputs from all the assets involved in the urban water cycle of the Poole Harbour Catchment.

Table 6.9.: Data inputs for the computation of the blue ($WF_{WTP,blue}$) and the grey ($WF_{WTP,grey}$) components of the $WF_{WTP,total}$.

Data input for the computation of the $WF_{WTP,blue}$	Unit	Arithmetic Figures
Energy Consumption	KWh/m^3	0.458
Coal (GAC treatment)	Kg/m^3	0.00262
Sodium Chloride (N-removal)	Kg/m^3	0.0243
Chlorine Gas (Cl-disinfection)	Kg/m^3	0.00039
Data or the computation of the $WF_{WTP,grey}$	**Unit**	**Figures**
Q_{in} BH1 [volume of water]	m^3/month	21,000
Q_{in} BH2 [volume of water]	m^3/month	12,000
C_{inlet} [total N]	gr/m^3	7.4
C_{nat} [total N]	gr/m^3	0.0
C_{max} [total N]	gr/m^3	50.0

Table 6.10.: Arithmetic Figures of the Water Footprints of the Urban Water Cycle (actor: Water Company) at a catchment scale.

Type of Water Footprint	Unit	Arithmetic Figures
$WF_{WTP,blue}$	m^3/month	$8.00 * 10^4$
$WF_{WTP,grey}$	m^3/month	$-2.81 * 10^5$
$WF_{WTP,total}$	m^3/year	$-2.41 * 10^6$
$\boldsymbol{WF_{WTP,catch}}$	m^3/year	$-4.34 * 10^7$
$WF_{WWTP,blue}$	m^3/month	$1.80 * 10^5$
$WF_{WWTP,grey}$	m^3/month	$5.39 * 10^5$
$WF_{WWTP,total}$	m^3/year	$8.63 * 10^6$
$\boldsymbol{WF_{WWTP,catch}}$	m^3/year	$1.99 * 10^8$
$\boldsymbol{WF_{urban,catch}}$	m^3/year	$\mathbf{1.55 * 10^8}$

6.3.3. Actor: Agriculture

The Water Footprint Assessment methodology (Hoekstra et al. 2011) was used to formulate the water inventory of the actor 'Agriculture'. This involved the estimation of water volumes consumed from the two sectors of this actor: crop cultivation and livestock production. Due to time and data constraints, a number of assumptions were made for the computations performed, based on literature or expert input from the project's partner.

For the crop cultivation sector, the total water footprint of the sector ($WF_{proc,crop}$) was computed. This would be equal to the sum of water footprint of all the relevant sectors and their processes occurring in the sector; thus, the sum of the water footprints of the sectors of Irrigation, Harvesting and Fertilising. However, for the needs of the research undertaken, it was assumed that the total water footprint of the sector 'crop cultivation' is equal to the components of the water footprints of the selected processes of the sectors of Irrigation and Fertilising. Due to data and time limitations, the water footprint of the sector 'Harvest' – which would include the total water incorporated into the harvested crops- was not computed. For the sector 'Irrigation' the processes of water storage and water transportation were not included in the computations. For the sector 'Fertilising' the grey water footprint alone was computed. Computations did not include the production and transportation phases of the fertilisers, assuming they take place outside of the catchment boundaries.

Thus, the following equation describes the rationale followed for the computation of the total water footprint of the activity 'crop cultivation' (as the sum of the previously analysed processes of the sectors Irrigation and Fertilising), $WF_{crop,total}$:

$$WF_{crop,total} = WF_{crop,proc,blue} + WF_{crop,proc,green} + WF_{crop,grey} \quad [volume/time] \ (21).$$

The blue ($WF_{crop,proc,blue}$) and then green ($WF_{crop,proc,green}$) components of the total water footprint were computed according to the following equations respectively:

$$WF_{crop,proc,blue} = \frac{CWU_{blue}}{Y} \quad [volume\ /\ mass] \ (22)$$

$$WF_{crop,proc,green} = \frac{CWU_{green}}{Y} \quad [volume\ /\ mass] \quad (23)$$

where

CWU is the crop water use (for blue and green water respectively) in m^3/ha

Y is the crop yield in tn/ha.

The blue and the green components of the crop water use (CWU) were calculated by the accumulation of daily evapotranspiration over the complete length of the growing period (lgp) of the individual crop:

$$CWU_{green} = 10x \sum_{d=1}^{lgp} ETgreen \;\; [volume \,/\, area] \; (24)$$

$$CWU_{blue} = 10x \sum_{d=1}^{lgp} ETblue \;\;\; [volume \,/\, area] \; (25)$$

where:
$ETgreen$, $ETbl$ue the green and the blue evapotranspiration respectively.

In equations (24) and (25), the factor 10 is used to convert water depths in millimetres into water volumes per land surface in $\frac{m^3}{ha}$. The 'green' crop water use represents the total rainwater evaporated from the field during the growing period; the 'blue' crop water use represents the total irrigation water evaporated from the field. The green and the blue water use during crop growth for the crops cultivated in the Poole Harbour Catchment were computed using the CROPWAT model (version 8.0; FAO 2010a). The CROPWAT model offers two different options to compute evapotranspiration: the 'crop water requirements' (CWR-assuming optimal growth conditions) and the 'irrigation schedule option'. For the case study presented, the latter option was selected, as a more accurate approach recommended by Hoekstra et al. (2011). The irrigation requirements (mm of water /growing period) were assumed to represent the $ETc, blue$. The component $ETc, green$ was computed as the difference between the total evapotranspiration of the crop minus its irrigation requirements: $ETc, green = ETc, crop - ETc, blue \;\; (26)$. Details on the model structure and methods of the CROPWAT model and a comprehensive manual for its use can be found online (FAO 2010a) and in the Water Footprint Assessment manual (Hoekstra et al. 2011).

For the performance of the ETc computations for each of the crops cultivated in the Poole Harbour Catchment (wheat, barley, maize, oilseed rape), data from the CLIMWAT database (version 2.0, FAO 2010b) were inputted in the CROPWAT model. The winter variations of wheat and oilseed rape were also included in the computations, as these crops are identified in literature (NRS report, EA, 2013) as the winter cover crops mainly cultivated in the Poole Harbour Catchment. The input of MORECS catchment data in the CROPWAT model was not possible due to format issues. CLIMWAT data from two meteorological stations located nearby the delineated area of the Poole Harbour Catchment were used, namely Exeter and Bournemouth, which belong to the MORECS squares 180 and 181 respectively. The catchment values were computed according to the equation (4) as introduced earlier. The length of growing periods of each of the crops included in the computations were inputted manually, based on available literature (**Figure 6.13**). The aggregated ETc computations for the main crops of the Poole Harbour Catchment are presented in **Table 6.11**. The detailed analysis of the data as produced by the CROPWAT model is presented in APPENDIX B.

For each of the crop types, ETc ($ETblue$ and $ETgreen$) were computed for both stations (Exeter and Bournemouth). Then, the arithmetic figures were aggregated at a catchment scale. The catchment values were then used to compute the blue (CWU_{blue}) and green (CWU_{green}) water use and the blue ($WF_{proc,crop,blue}$) and the green $\left(WF_{proc,crop,green}\right)$ water footprints of each crop type. Due to the lack of catchment-specific data for crop yields, it was assumed that the UK average yield data – retrieved from DEFRA National Statistics (2015) and Nix (2015)- apply to the Poole Harbour Catchment. Then, the annual catchment yield (tn) of each of the crops was computed by multiplying their land cover (ha) with the average crop yield (tn/ha). The total (blue and green) water consumed per crop type at a catchment scale on an annual basis is then computed by multiplying the water footprint values with the catchment annual yield.

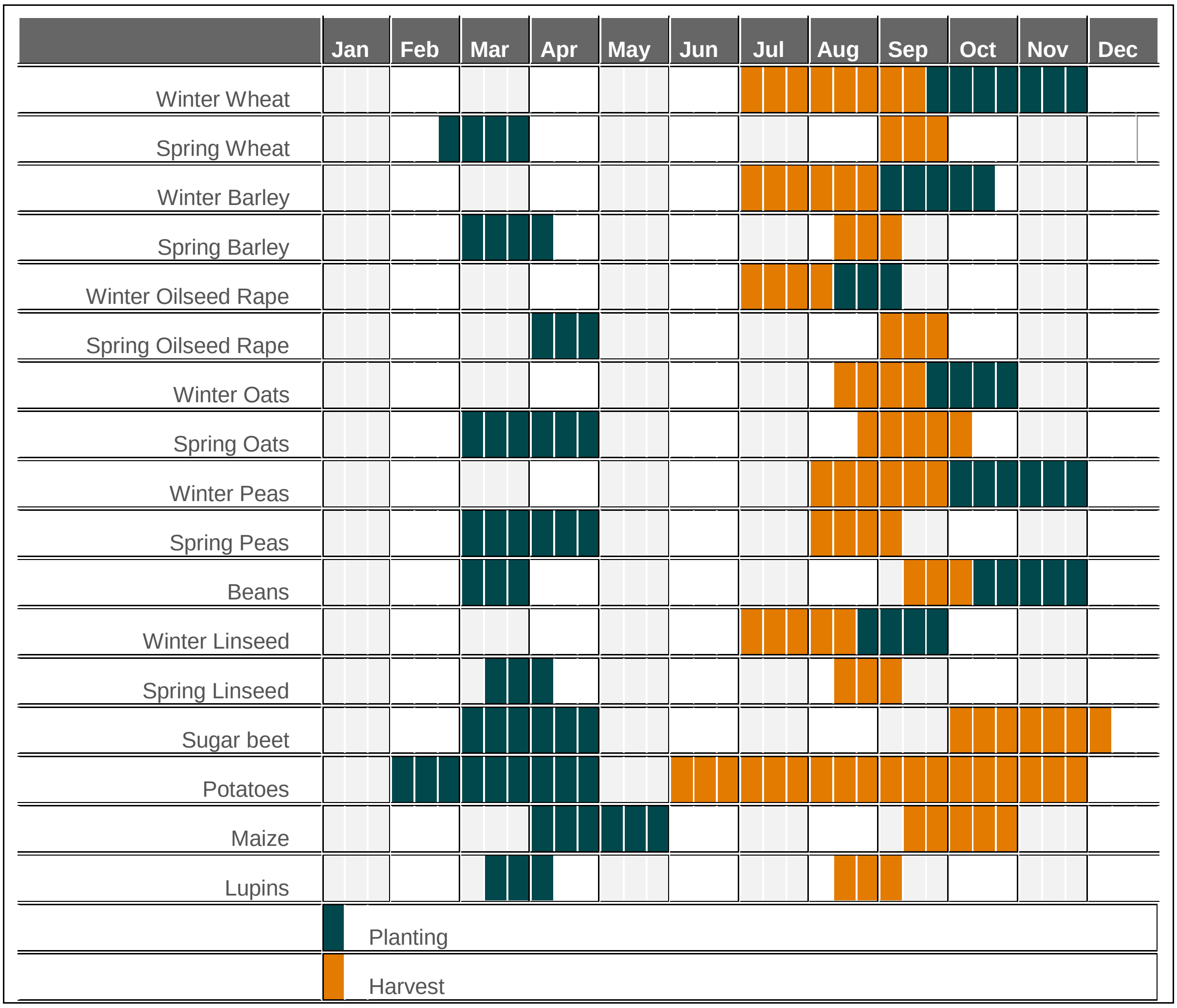


Figure 6.13.: Crop calendar for main UK crops. Abstracted from *www.criddle.co.uk.* Visited on 1st of August 2016.

The grey component of the water footprint of the process of 'growing a crop', $WF_{crop,grey}$, was based on the method introduced from Hoekstra et al. (2011). According to their approach, the $WF_{crop,grey}$ is computed as follows:

$$WF_{crop,grey} = \frac{(axAR)/(c_{max} - c_{nat})}{Y} \quad [volume/mass] \ (27)$$

where:
a is the leaching run-off fraction (%)
AR is the chemical application rate to the field (kg/ha)
c_{max} is the maximum acceptable concentration in water bodies (mg/l) for the pollutant considered
c_{nat} is the natural concentration for the pollutant considered (mg/l)
Y is the crop yield (ton/ha).

For each of the crops cultivated in the Poole Harbour Catchment, the grey components of the water footprint of the process of growing a crop, $WF_{crop,grey}$, was computed, making use of diverse data sources (**Table 6.12.; Table 6.13.**). Those included literature (e.g. Mekonnen and Hoekstra 2015; Nix 2015; Hoekstra et al. 2011), past studies (EA and WFN 2014; EA, 2013), online sources and expert input from agronomists working with the project's industrial partner. The experts mainly advised on the performance of measures implemented for the reduction of total nitrogen in the Poole Harbour catchment and the formulation of assumptions regarding chemical application and leaching run-off rates.

For the sector 'livestock', an aggregated water footprint approach was followed, due to data limitations. The blue and the green water footprints of the sector 'livestock' were not computed. The grey component of the water footprint of the sector, $WF_{live,grey}$, was calculated as the difference between the water footprint of the sector 'growing a crop' from the total water footprint of the stakeholder 'agriculture' which stands for the total nitrogen load (tn/year) from diffuse sources polluting the Poole Harbour. Data were retrieved from previous studies (NRS, EA, 2013), describing both the current status of the catchment/harbour and future environmental goals (**Table 6.12.**).

For the calculation of the grey water footprint of the stakeholder 'agriculture', $WF_{agri,grey}$, a simplified version of the grey water footprint was applied, based on Hoekstra et al. (2011):

$$WF_{agri,grey} = \frac{L}{c_{max} - c_{nat}} \quad [volume/time] \quad (28)$$

where:
L the load of total N in the Poole Harbour (kg)
c_{max} the maximum acceptable concentration of total N in the harbour (mg/l)
c_{nat} is the natural concentration of total N in the harbour (mg/l).

Consequently, the grey water footprint of the sector 'livestock', $WF_{live,grey}$, was computed as follows:

$$WF_{live,grey} = WF_{agri,grey} - WF_{crop,grey} \quad [volume/time] \text{ (29).}$$

Table 6.11: CROPWAT model results for the crops located in the Poole Harbour Catchment. Stations Exeter (EXE) and Bournemouth (BOU) were assumed to represent MORECS squares 180 and 181 respectively.

Crop Water Use and Blue & Green Water Footprints in the Poole Harbour Catchment										
Crop type	**ET_{blue} (mm)**		**$ET_{c,catch,blue}$ (mm)**	**ET_{green} (mm)**		**$ET_{c,catch,green}$ (mm)**	**$CWU_{blue,catch}$ (m^3/ha)**	**WFc,blue (m^3/tn)**	**$CWU_{green,catch}$ (m^3/ha)**	**WFc,green**
	EXE	**BOU**	***catchment***	**EXE**	**BOU**	***catchment***	***catchment***	***catchment***	***catchment***	***catchment***
Spring Wheat	196.5	117.2	158.4	175.7	163.0	169.6	1584.4	176.0	1696.0	188.4
Barley	192.5	108.7	152.3	172.6	161.7	167.4	1522.8	220.7	1674.0	242.6
Maize	251.6	1559.1	207.2	172.4	162.5	167.6	2072.0	276.3	1676.0	223.5
Spring Oilseed rape	167.8	97.3	134.0	153.1	147.6	150.5	1339.6	343.5	1505.0	385.8
Winter Wheat	363.5	235	301.8	351.9	306.9	330.3	3018.2	335.4	3303.0	367.0
Winter Oilseed rape	44.9	10.8	28.5	167.2	147.9	157.9	285.3	73.2	1579.0	405.0

Table 6.12: Yield data from (a) National Farming Statistics, DEFRA 2015 and (b) Farm Management Pocketbook, Nix 2015. Land cover data from NRS (EA,2013) report. The Catchment Annual Yield (tn) is computed per crop as the product of the proliferation of the average crop yield by catchment land cover. Scenarios: (1) current status, including only spring crops; (2) implementation of winter cover crops without starter fertiliser (20kg/ha).

Crops, Yields (ton/ha), Fertiliser (N) application rates and Grey Water Footprints in the Poole Harbour Catchment									
Crop type	**Average Crop Yield (tn/ha)**	**Catchment Land Cover (ha)**	**Catchment Annual Yield (tn)**	**N application (kg/ha)[1]**	**N application (kg/ha)[2]**	**WFc,grey (m^3/tn) [1]**	**WFc,grey (m^3/tn) [2]**	**Other parameters**	
								Concentration of total N	
Spring Wheat	9.0[(a)]	9000	$81*10^3$	150	130	2000.0	577.8	C_{max}=2.9mg/l	C_{nat}=0.4 mg/l
Barley	6.9 [(b)]	1750	$12.1*10^3$	160	140	2782.6	811.6	**Leaching Run-off Factor α**	
Maize	7.5 [(b)]	1750	$13.1*10^3$	200	180	3200.0	960.0	[1:] α =30%	[2:] α =10%
Spring Oilseed rape	3.9 [(a)]	1750	$6.8*10^3$	80	60	2461.5	615.4	**Aggregated WFs**	
Winter Wheat	9.0[(a)]	9000	$81*10^3$	-	230	-	1022.2	**WFgrey,agri (m^3/year):** $7.8*10^8$	
Winter Oilseed rape	3.9 [(a)]	1750	$6.8*10^3$	-	170	-	1743.6	**WFgrey,live (m^3/year):** $5.3*10^8$	

Table 6.13: Water Consumption per crop at a catchment scale. Scenarios: (1) current status, including only spring crops; (2) implementation of winter cover crops without starter fertiliser (20kg/ha).

Crop type	**Blue Water (m^3)**	**Green Water (m^3)**	**Grey Water[1] (m^3)**	**Grey Water[2] (m^3)**	**Volume of Water (m3)**
Spring Wheat	$1.4*10^7$	$1.5*10^7$	$1.6*10^8$	$4.7*10^7$	[1] $3.8*10^8$
Barley	$2.7*10^6$	$2.9*10^6$	$3.4*10^7$	$9.8*10^6$	
Maize	$3.6*10^6$	$2.9*10^6$	$4.2*10^7$	$1.3*10^7$	
Spring Oilseed rape	$2.3*10^6$	$2.6*10^6$	$1.7*10^7$	$4.2*10^6$	[2] $2.7*10^8$
Winter Wheat	$2.7*10^7$	$3.0*10^7$	-	$8.3*10^7$	
Winter Oilseed rape	$5.0*10^5$	$2.8*10^6$	-	$1.2*10^7$	

6.4. Water Accounts and the Catchment Physical Input-Output Table

This section illustrates how the methodologies introduced for the individual actors of the catchment and the indices selected to show the outputs of their processes, feed into the population of the Catchment Physical Input Output Table (Catchment PIOT).

Table 6.14. presents the updated Catchment PIOT, showing which cells are filled with the hydrology figures, water footprint indices or other physical figures (e.g. green water requirements or leakage water). Based on the assumptions made throughout the research and simplifications forced due to methodological limitations, the Catchment PIOT could be restructured to represent more aggregated sectors, as shown in **Table 6.15**. In this version, the number of empty cells is limited, as the available in literature indices enable to compute the outputs of each of the combinations between sectors.

The structure of the Catchment PIOT rules the rationale of its population with outputs of the water inventories. Thus, the column indicates the 'From' and the rows indicate the 'To'. For example, for filing in the X(1,1) cell, showing the water circulation within the Atmosphere, the question asked is: "How much the sector *Atmosphere* contributes to itself?". The hydrological figure of evapotranspiration, ET_{atm}, describes this relationship. Based on this rationale, the figure of $WF_{blue,catch,WTP}$ describes "how much the sector *Hydrosphere* contributes to the urban water cycle", while the figure of $WF_{grey,catch,WTP}$ describes "how much the urban water cycle contributes to the sector *Hydrosphere*'.

The questions asked for filling in the Catchment PIOT are dictated by the System Dynamics model created for the catchment system under study. Thus, the indices or hydrological figures populating the table, map back to the relationships identified in a previous research stage. This enables the parallel development of the modelling, cross-check of results and adaptations or corrections when necessary.

Table 6.14.: The Catchment PIOT as formulated according to the methodologies presented for each of the catchment's actors. The indices introduced in this section are used to populate the matrix. The columns represent the 'From' and the rows the 'To'. The indices show the volume of water utilised from each sector: for example, the WFcatch,WTP represents the volume of water abstracted from the sector 'Hydrosphere' and the WFWWTPcatch represents the volume of water returned to the sector 'Hydrosphere' .The volumes of water refer to cubic meters on an annual basis (m^3/year).

	Atm	***Hydro***	***Pedo***	***Litho***	***Abstr***	***W-Treat***	***W-Distrib***	***WW-Distrib***	***WW-Treat***	***Irrig***	***Harv***	***Fertil***	***W-Anim***	***Feed***
Atmosphere	ET_{atm}	R	*Soil moisture*	I	-	-	-	-	-	-	-	-	-	-
Hydrosphere	ET_{blue}	EFR	-	-	$WF_{blue,catch,WTP}$		$WF_{blue,catch,distrib}$		$WF_{blue,catch,WWTP}$	$WF_{blue,irr}$	$WF_{blue,harv}$	$WF_{blue,virt,ft}$	$WF_{blue,an}$	$WF_{blue,virt,fe}$
Pedosphere	ET_{green}	-	*Green WR*	-	$WF_{green,catch,WTP}$		$WF_{green,catch,distrib}$		$WF_{green,catch,WWTP}$	$WF_{green,irr}$	-	-	-	-
Lithosphere	-	*Ground water*	-	*Stock water*							$WF_{green,harv}$	$WF_{gr,virt,ft}$	$WF_{green,an}$	$WF_{gr,,virt,fe}$
Abstraction	$ET_{t,}$	$WF_{grey,catch,WTP}$	-	-	-	-	-	-	-	-	-	-	-	-
Water Treatment	$ET_{t,urb,2}$		-	-	*Water Flows within the Urban Water Cycle & Virtual Water*					-	-	-	-	-
Water Distribution	$ET_{t,urb,3}$	$WF_{grey,catch,distrib}$	-	*leakage*						-	-	-	-	-
Wastewater Distribution	$ET_{t,urb,4}$		-	*leakage*						-	-	-	-	-
Wastewater Treatment	$ET_{t,urb,5}$	$WF_{grey,catch,WWTP}$	-	-						-	-	-	-	-
Irrigation	$ET_{t,irrig}$	-	-	-	-	-	-	-	-	*Water Flows & Virtual Water*				
Harvest	-	-	-	-	-	-	-	-	-					
Fertilising	-	$WF_{grey,ft}$	$WF_{grey,ft}$	$WF_{grey,ft}$	-	-	-	-	-					
Watering Animals	-	$WF_{grey,an}$	$WF_{grey,an}$	$WF_{grey,an}$	-	-	-	-	-					
Feed	-	-	-	-	-	-	-	-	-					

Table 6.15.: Re-structured catchment PIOT due to methodological assumptions and limitations. All figures refer to m³/year.

	Atmosphere	*Hydrosphere*	*Pedosphere*	*Lithosphere*	*Urban Water Cycle*	*Crop Cultivation*	*Livestock*
Atmosphere	ET_{atm}	R	$Soil\ moisture$	I	-	-	-
Hydrosphere	ET_{blue}	EFR	-	-	$WF_{blue,catch,urban}$	$WF_{blue,crop}$	$WF_{blue,liv}$
Pedosphere	ET_{green}	-	$Green\ WR$	-	-	$WF_{green,crop}$	-
Lithosphere	-	$Ground\ water$	-	$Stock\ water$	$WF_{green,catch,urban}$		$WF_{green,liv}$
Urban Water Cycle	ET_{urban}	$WF_{grey,catch,urban}$			-	-	-
Crop Cultivation	ET_{crop}	$WF_{grey,crop}$			-	-	-
Livestock	ET_{liv}	$WF_{grey,liv}$			-	-	-

Table 6.16.: Scenarios. *assuming efficiency of WWTP at 7mg/l.

Land Management Scenarios	**Benchmark (N load, tn/year)**	**Reduction Target (N load, tn/year)**	**Source of Pollution**	
			Diffuse	**Point ***
Current status (S0)	2280	-	1950	330
Winter Cover Crops (S1)	1730	550	1400	330
Site-specific management (S2)	1730	550	1400	330

6.5. The application of the Catchment Metabolism in a scenario analysis

This section presents the use of the complete modelling approach. It synthesises the outputs of the previous chapters and sections in order to apply the Catchment Metabolism and its multiple water inventories for the Poole Harbour Catchment and for a number of scenarios in regards to tackling nitrogen pollution.

A scenario analysis is firstly introduced. Then, a Catchment Physical Input-Output Table (Catchment PIOT) is produced for each of the scenarios. Due to the nature of the Catchment PIOT, only the outputs of the interactions among actors are presented in the final PIOT, which is, essentially, a portfolio presentation of environmental outputs. Selected indices from environmental impact assessment methodologies are utilised to compute the environmental outputs. The underpinning methodology for these computations is also discussed, whilst the numeric hydrological analysis performed as part of the actors' water inventories provides the hydrometric values used for the calculations.

6.5.1. Scenario Formulation

The SD model of the Poole Harbour Catchment (**Figure 6.9**.) has revealed the synergies between the biosphere and the technosphere within the Poole Harbour Catchment, and there is evidence (NRS, EA, 2013; Chapter 4) showing that a joint tackle of both point and diffuse nitrogen pollution would prove more beneficial for its overall environmental performance.

A scenario analysis has been performed for the identification of different strategies applied to the Poole Harbour Catchment. The aim of each of the strategies is to enable the system to meet the statutory standards for water quality of the final recipient (i.e. the Poole Harbour) in the most sustainable means. Therefore, the overall environmental performance of the catchment system is assessed for each of the scenarios formulated.

The current environmental performance of the Poole Harbour Catchment is firstly assessed and then used as the baseline scenario (S0). Then, two scenarios are formulated (S1, S2) based on the nitrogen reduction targets set in previous studies (Nitrogen Reduction Strategy, EA, 2013) (**Table 6.16.**). A 'reverse engineering' approach is used for the assessment of the strategies. That is, each of the scenarios assumes the same final target in terms of the overall nitrogen reduction and then assesses how this is achieved through the implemented strategy. The strategies are not, therefore assessed for their effectiveness; they are rather assessed for their environmental performance, assuming that the nitrogen reduction targets are met. The application of the Catchment

Metabolism modelling schema for each of the strategies serves this assessment, as it reveals the trade-offs among the subsystems or actors of the catchment system. This is also a means of evaluating the modelling schema per se for its suitability to provide structured and uniform evidence for informing holistic, catchment-based asset management and planning.

The overall environmental target in the Poole Harbour Catchment is the reduction on the nitrogen load deriving from diffuse pollution by approx. 550 tonnes of total N across the catchment at an annual basis ($\mathrm{R_{benchmark}} = 550\ \mathrm{tn/\ year}$). The two scenarios formulated (S1, S2) represent two different agricultural practices as recommended by previous studies and catchment's stakeholders (Nitrogen Reduction Strategy, EA, 2013). These include the establishment of winter crops (wheat winter and winter oilseed rape production) and the adoption of site-specific management along the catchment (scenarios S1 and S2 respectively. Different fertiliser application rates and modelling parameters are assumed in the scenarios formulated, based on literature and expert input from the project's industrial partner. Thus, the fertiliser (total N) application rate for the winter cover crops scenario (S1) assumes that the starter fertiliser (20 Kg N/ha) is not applied, as it is captured in the soil from the previous crop rotation. The leaching run-off factor α is assumed to be differentiated within seasons (e.g. Hardie et al. 2012; Alberts et al. 1978), as it increases during spring, especially after bear soils (no winter cover crops) during winter. For the case study undertaken, it is assumed $\alpha = 10\%$ (Chapagain et al. 2006) for scenario S1 and $\alpha = 30\%$ (expert input) for scenarios S0 and S2. Both scenarios assume that the contribution of point sources (urban water cycle, through the wastewater treatment plants) to the total nitrogen load do not exceed 15% at an annual basis ($\mathrm{P_{point}} = 330\ \mathrm{tn/year}$). The operation efficiency of the nitrogen removal plants is also assumed stable ($\mathrm{Urban_{efficiency}} = 7\mathrm{mg\ N/l}$).

A set of rules were defined, aiming to ensure a rigorous comparison of the environmental performance of the scenarios. For all the scenarios, the environmental outputs are computed over the period of a hydrological year. The re-structured Catchment PIOT and the water inventories underpinning its formulation are used for all the scenarios formulated. The assumptions and limitations apply are as described in previous sections of the chapter. The overall environmental performance of each of the scenarios is analysed as compared with the current status of the catchment (S0). The construction of Catchment PIOTs for the wet and the dry periods were not included in the analysis, as this step was considered out of the scope of the research. Nonetheless, seasonality was included in the hydrological analysis of the catchment system.

6.5.2. Implementation of the Catchment PIOTs & Analysis of the results

Following the formulation of the scenarios for tackling nitrogen pollution in the Poole Harbour Catchment, a Catchment PIOT is constructed for each of them (**Table 6.17.**; **Table 6.18.**; **Table 6.19.**). For facilitating the comparison among scenarios, a pictorial representation (Sankey diagram) of the results follows (**Figure 6.14.; Figure 6.15.; Figure 6.16.**). The section then analyses the results of the implementation of the Catchment PIOTs for the assessment of the environmental performance of the scenarios formulated for reducing nitrogen pollution in the Poole Harbour Catchment.

The construction of the Catchment PIOTs is based on the rules and methodologies formulated earlier in this chapter. The outputs depicted allow for the comparison of the environmental performance of the scenarios, against the current status of the catchment system and between them. For the description of the current status (S0) and for both the scenarios formulated (S1, S2), the optimum status of the river systems was assumed; thus, the environmental flow requirements (EFR) were calculated for that status.

Despite the identification of the physical water flows between several sectors of the catchment system, a number of outputs were not calculated for none of the scenarios. For example, *soil moisture*, as the outputs of the sector 'Atmosphere' to the sector 'Pedosphere' or the blue ($WF_{blue,liv}$) and the green ($WF_{green,liv}$)water footprints of the sector 'livestock', as the output of the sector 'Hydrosphere'. This is due to either simplifications made in the formulation of the scenarios or lack of catchment-specific models or data for the computation of the physical arithmetic values. A colour-scale is used to justify the uncertainty of the data used or processed for the Poole Harbour Catchment case study. As such, *Red* represents high uncertainty, *Yellow* represents potential uncertainty and *Green* represents confidence in the data.

The non-computed figures represent the following physical flows or stocks: *soil moisture:* green water captured in the pedosphere; *green WR*: green water requirements, thus: the green water needed to maintain the biodiversity of the pedosphere; *groundwater*: the volume of groundwater transferred from the saturated aquifer to the river; *stock*: the groundwater stored in the aquifer at an annual rate;ET_{urban}: the volume of water evaporated during the urban water cycle, assuming is not considered as a closed system; ET_{liv}: the volume of water evapo-transpirated from the sector of 'livestock'.

Table 6.17.: Catchment PIOT for S0: current status of the Poole Harbour Catchment. Values present volume of water per hydrological year (m^3/year). Due to the number of assumptions and multiple sources of data, there is limited confidence in the arithmetic values presented.

<table>
<tr><th></th><th>Atmosphere</th><th>Hydrosphere</th><th>Pedosphere</th><th>Lithosphere</th><th>Urban Water Cycle</th><th>Crop Cultivation</th><th>Livestock</th></tr>
<tr><td>Atmosphere</td><td>$4.11 * 10^8$</td><td>$2.88 * 10^8$</td><td>$Soil\ moisture$</td><td>$2.36 * 10^7$</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td>Hydrosphere</td><td>$2.90 * 10^8$</td><td>$2.30 * 10^8$</td><td>-</td><td>-</td><td>$6.70 * 10^7$</td><td>$2.30 * 10^7$</td><td>$WF_{blue,liv}$</td></tr>
<tr><td>Pedosphere</td><td>$1.23 * 10^8$</td><td>-</td><td>$Green\ WR$</td><td>-</td><td rowspan="2">$WF_{green,catch,urban}$</td><td rowspan="2">$2.40 * 10^7$</td><td rowspan="2">$WF_{green,liv}$</td></tr>
<tr><td>Lithosphere</td><td>-</td><td>$Ground\ water$</td><td>-</td><td>$Stock\ water$</td></tr>
<tr><td>Urban Water Cycle</td><td>ET_{urban}</td><td colspan="3">$8.81 * 10^7$</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td>Crop Cultivation</td><td>$1.07 * 10^6$</td><td colspan="3">$2.50 * 10^8$</td><td>-</td><td>-</td><td>-</td></tr>
<tr><td>Livestock</td><td>ET_{liv}</td><td colspan="3">$5.26 * 10^8$</td><td>-</td><td>-</td><td>-</td></tr>
</table>

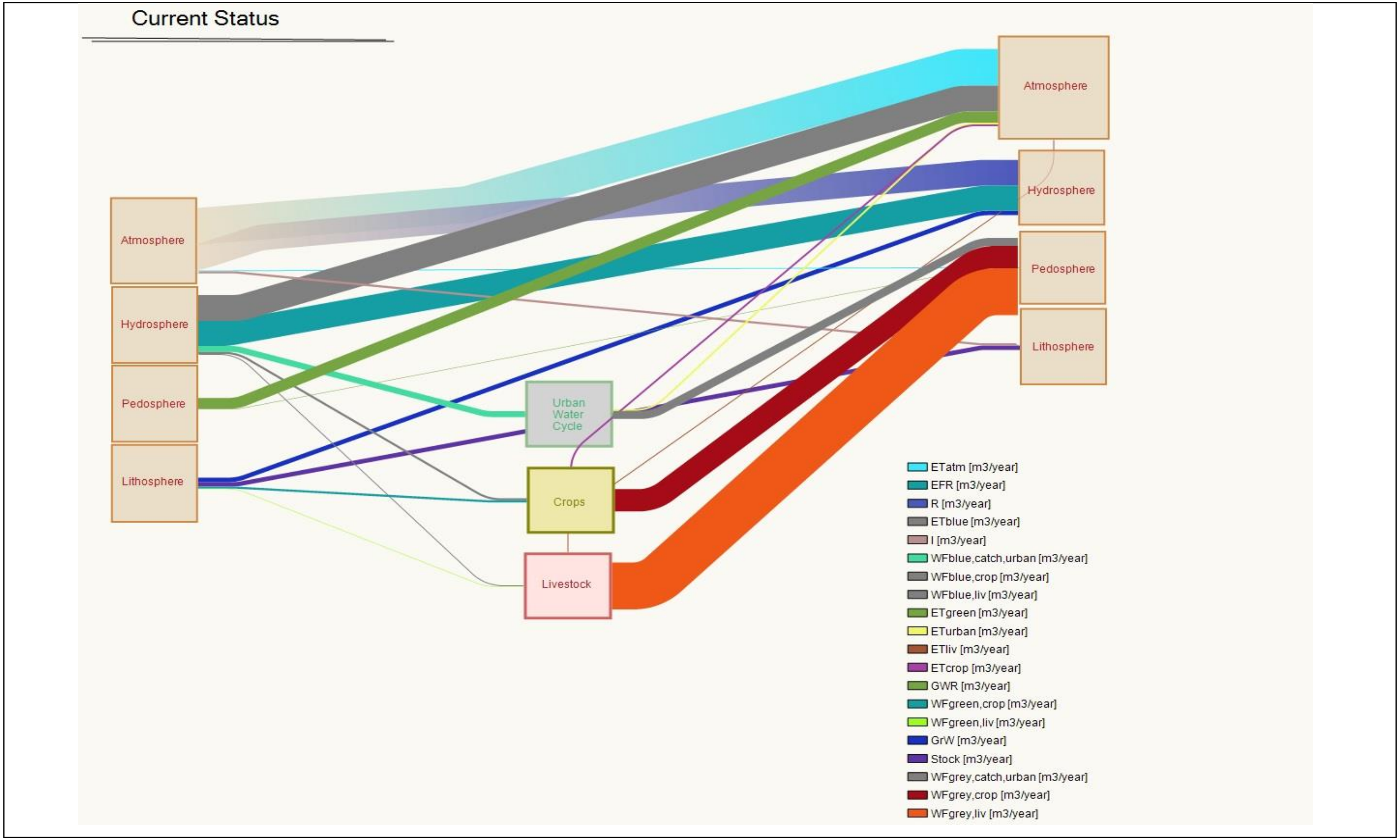


Figure 6.14.: Sankey diagram describing the flow exchange for the current status of the Poole Harbour Catchment (scenario S0).

Table 6.18.: Catchment PIOT for S1: implementation of winter cover crops across the Poole Harbour Catchment. The scenario assumes the non-application of the starter fertiliser (20 Kg N/ha) across the catchment, for all crops considered. Values present volume of water per hydrological year (m^3/year). Due to the number of assumptions and multiple sources of data, there is limited confidence in the arithmetic values presented.

	Atmosphere	***Hydrosphere***	***Pedosphere***	***Lithosphere***	***Urban Water Cycle***	***Crop Cultivation***	***Livestock***
Atmosphere	$4.11 * 10^8$	$2.88 * 10^8$	$Soil\ moisture$	$2.36 * 10^7$	-	-	-
Hydrosphere	$2.90 * 10^8$	$2.30 * 10^8$	-	-	$6.70 * 10^7$	$5.10 * 10^7$	$WF_{blue,liv}$
Pedosphere	$1.23 * 10^8$	-	$Green\ WR$	-	$WF_{green,catch,urban}$	$5.60 * 10^7$	$WF_{green,liv}$
Lithosphere	-	$Ground\ water$	-	$Stock\ water$			
Urban Water Cycle	ET_{urban}	$8.81 * 10^7$			-	-	-
Crop Cultivation	$1.74 * 10^6$	$1.70 * 10^8$			-	-	-
Livestock	ET_{liv}	$3.92 * 10^8$			-	-	-

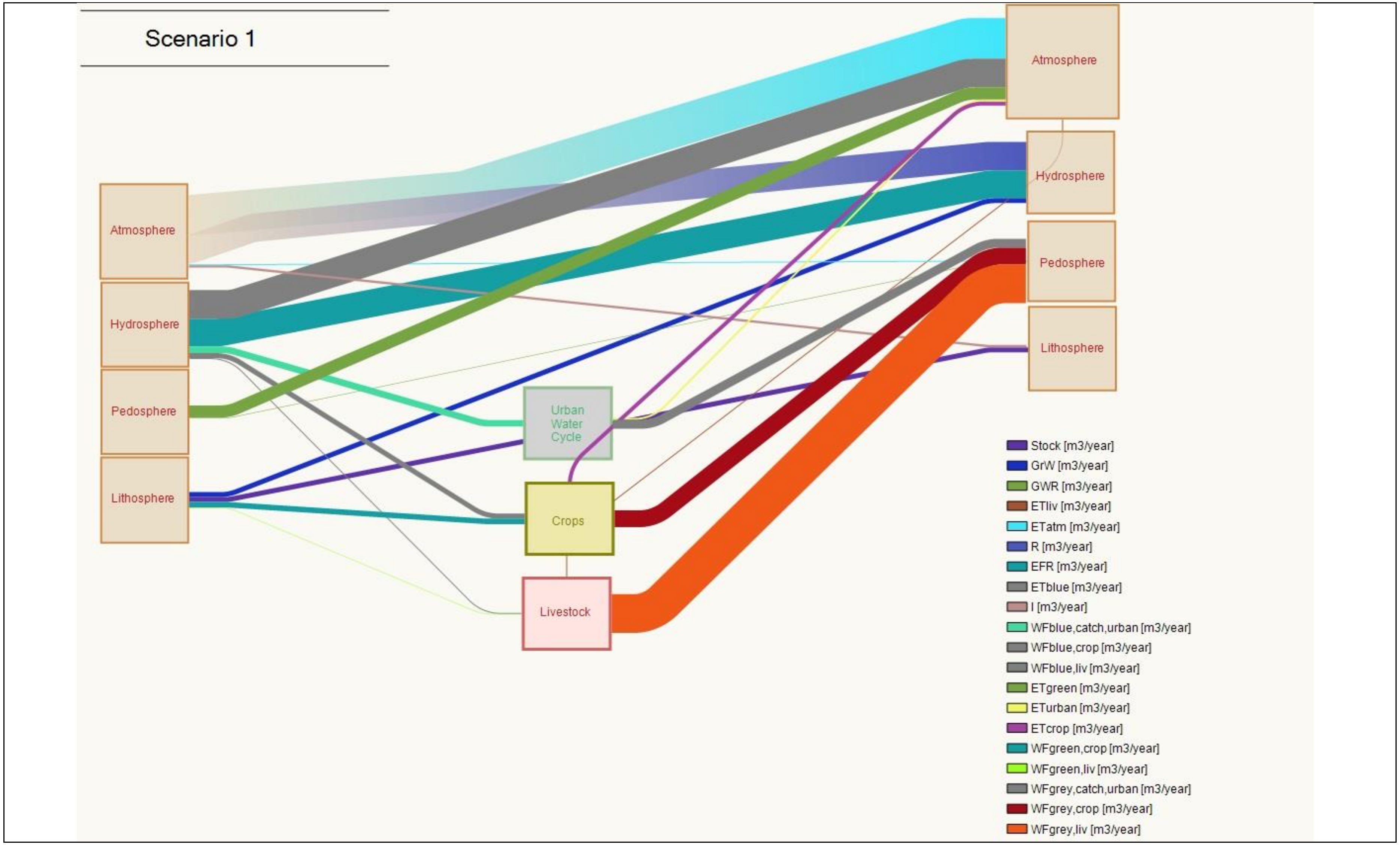


Figure 6.15.: Sankey diagram describing the flow exchange for the implementation of winter cover crops across the Poole Harbour Catchment (scenario S1).

Table 6.19.: Catchment PIOT for S2: implementation of precision agriculture across the Poole Harbour Catchment. The scenario assumes the non-application of starter fertiliser (20 Kg N/ha) across the catchment, but not the implementation of winter cover crops. The leaching run-off factor (α) is, thus, higher than in S1. Values present volume of water per hydrological year (m^3/year). Due to the number of assumptions and multiple sources of data, there is limited confidence in the arithmetic values presented.

	Atmosphere	***Hydrosphere***	***Pedosphere***	***Lithosphere***	***Urban Water Cycle***	***Crop Cultivation***	***Livestock***
Atmosphere	$4.11 * 10^8$	$2.88 * 10^8$	$Soil\ moisture$	$2.36 * 10^7$	-	-	-
Hydrosphere	$2.90 * 10^8$	$2.30 * 10^8$	-	-	$6.70 * 10^7$	$2.30 * 10^7$	$WF_{blue,liv}$
Pedosphere	$1.23 * 10^8$	-	$Green\ WR$	-	$WF_{green,catch,urban}$	$2.40 * 10^7$	$WF_{green,liv}$
Lithosphere	-	$Ground\ water$	-	$Stock\ water$			
Urban Water Cycle	ET_{urban}	$8.81 * 10^7$			-	-	-
Crop Cultivation	$1.07 * 10^6$	$2.20 * 10^8$			-	-	-
Livestock	ET_{liv}	$3.40 * 10^8$			-	-	-

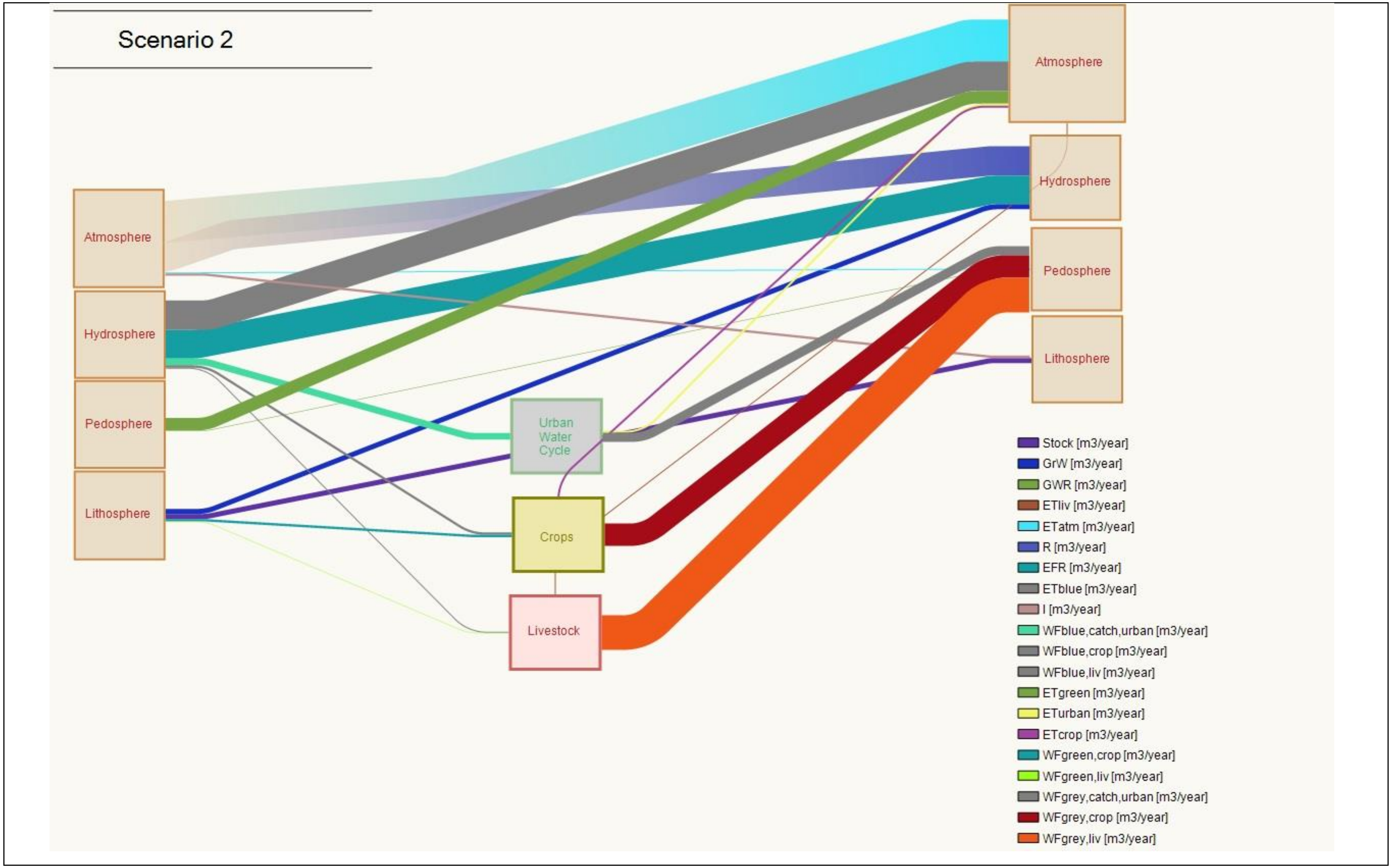


Figure 6.16.: Sankey diagram describing the flow exchange for the implementation of precision agriculture across the Poole Harbour Catchment (scenario S2).

The results show that the optimum strategy for the reduction of the nitrogen load in the Poole Harbour is the implementation of winter cover crops across the catchment (scenario S1). This strategy involves the overall reduction of fertiliser application in the catchment, as it assumes that the starter fertiliser needed for each of the cultivated crops (20 Kg N/had) is captured in the pedosphere due to the continuous crop circulation. Further, the leaching run-off is reduced (α=10%) which may result in non-quantified benefits for the catchment system, such as the reduction of river sedimentation due to limited erosion during winter / wet season. Compared to the implementation of precision agriculture across the catchment (scenario S2), strategy S1 shows a slightly higher grey water footprint of the sector 'livestock', which would disqualify S1 from 'optimum'. Nonetheless, the difference is rather negligible, when compared to the total nitrogen load or the total grey water footprint of the sector. The simplifications and assumptions made for the formulations of the scenarios do not allow for the detailed comparison or analysis of the contribution of livestock in the total nitrogen load of the Poole Harbor. The scenarios formulated did not consider improvements or changes in the management of the sector 'livestock' due to lack of data or recommendations from previous studies conducted from the project's industrial partner or third parties (e.g. environmental regulators, consultancy).

6.6. An alternative version of the Catchment PIOT

Hydrological parameters and Water Footprint figures have been used to populate the Catchment PIOT (**Table 6.14.**). Nevertheless, it that form, the Catchment PIOT does not show environmental impacts at a catchment scale. It rather shows the volume (e.g. runoff) or appropriation (e.g. blue water footprint) of the available freshwater used or consumed by the catchment's sectors or their individual sectors and their activities.

An alternative version of the Catchment PIOT (**Table 6.20.**) could be produced, aiming to integrate indicators or figures for mid-point impacts. This alternative option would relate the figures of the PIOT with elements such as Water Availability (WA) and Water Stress (WS). Moreover, it could integrate the indices from or related to the Water Footprint Assessment methodology, such as Water Pollution Level (WPL) (Hoekstra et al. 2011) and the Quantity-Quality-Environmental Flow Requirements (QQE) indicator (Liu et al. 2016). This expansion or alternative of the format of the Catchment PIOT would enable further environmental and hydrological analysis of the catchment system. It would also enhance the communication of complex environmental issues to non-experts, thanks to the aggregated information provided from the use of indicators. **Table 6.20.** shows the alternative version of the Catchment PIOT. The indexes describing potential mid-point, regional, water-related impacts are shown in the table, followed by the relevant academic reference. Due to the limitations of the available indices and the lack of catchment-specific data, computations are not performed for the Poole Harbour Catchment (i.e. the case study of the undertaken research). It should be noted that the figures describing water volumes relevant to the natural water cycle remain intact. The main differences are observed in the figures describing the relationships between the other actors of the catchment system (e.g. water company and agriculture). Multiple options for the use of indices are displayed for those inter-sectoral relationships or impacts which have been substantially discussed in existing literature.

Table 6.20.: Alternative Catchment PIOT. This version shows the use of the available indicators for potential mid-point, regional, water-related impacts. Different indicators are displayed for the inter-sectoral interactions which may be described by more than one indices.

	Atmosphere	***Hydrosphere***	***Pedosphere***	***Lithosphere***	***Urban Water Cycle***	***Crop Cultivation***	***Livestock***
Atmosphere	ET_{atm}	R	$Soil\ moisture$	I	-	-	-
Hydrosphere	ET_{blue}	EFR	-	-	$WF_{blue,catch,urban}$	$WF_{blue,crop}$	$WF_{blue,liv}$
Pedosphere	ET_{green}	-	$Green\ WR$	-	-	$WF_{green,crop}$	-
Lithosphere	-	$Ground\ water$	-	$Stock\ water$	$WF_{green,catch,urban}$		$WF_{green,liv}$
Urban Water Cycle	ET_{urban}	WPL or QQE	WPL or QQE	WScI or VI	WSI	WSI	WSI
Crop Cultivation	ET_{crop}	WPL or QQE	WPL or QQE	WScI or VI	WSI	WSI	WSI
Livestock	ET_{liv}	WPL or QQE	WPL or QQE	WScI or VI	WSI	WSI	WSI

WPL: Water Pollution Level (WPL) (Hoekstra et al. 2011); ***QQE***: Quantity-Quality-Environmental Flow Requirements indicator (Liu et al. 2016); ***WScI***: Water Scarcity Index (Döll 2009); ***VI***: Vulnerability Index (Döll 2009); ***WSI***: Water Stress Index (Pfister et al. 2009).

6.7. Concluding remarks on the Water Accounts of the Catchment PIOT

The section discusses the challenges and limitations of the water accounting methodologies, underpinning the population of the Catchment PIOT with indexes and data.

The strengths are summarised in the following:

- The use of established accounting methods in conjunction with Systems Dynamics modelling introduces a novel perspective in their joint use for Integrated Catchment Management.
- The creation of transparent methodologies for the water inventories of the catchment's actors enables detailed modelling of water processes at a catchment scale. This then leads to catchment-specific computations of environmental outputs and the assessment of the overall performance of a catchment system. The transparency of the flow mapping and accounting, contributes to sustainable integrated catchment management options, which are targeted to up-stream solutions, following on the principles that "*you cannot profoundly alter a system's outputs (i.e. wastes, emissions) without changing also its inputs and the ways it works internally*" (Fischer-Kowalski 2003). It also enables identifying research gaps, such as the type of hydrological models which need to be develop in order to describe the outputs among sectors or the water accounting or impact assessment indices missing from current literature.
- The use of the matrix representation creates a new approach to communicate the natural hydrological cycle to non-experts. The natural water budget is presented into a tabular, unified format, compatible with other fields of science, such as economics. The compartments of the natural water budget are depicted as outputs of processes among sectors: for example, the surface run-off is demonstrated as the output of the sector *Atmosphere* to the sector *Hydrosphere* and the Environmental Flow Requirements (EFR) are described as the contribution of the sector *Hydrosphere* to itself; thus, to the riverine ecosystem. This format also enables the identification of gaps on data. For example, for the Poole Harbour Catchment case study, the lack of data for the soil moisture does not allow for the computation of the outputs between the 'sectors' of pedosphere and lithosphere.

On the other hand, the water accounting methodologies suffer from a few methodological limitations, which mainly relate to the assumptions made for the selected catchment case study:

- The scenarios formulated to evaluate the practical value of the methodology did not include livestock management options. Thus, the comparison between the scenarios is not robust enough to provide ground for decision-making for the catchment system.

- In regards to the water accounting of the sector 'cultivating crops', the use of CLIMAT data in the CROPWAT model introduces an uncertainty in the results. This is mainly because CROPWAT assumes that the Effective Precipitation equals to the 80% of rainfall. These figures are not true for the Poole Harbour Catchment, where the Effective Precipitation was computed to be a 43% of the catchment's annual water budget.
- The distinction between green and blue water use is more relevant to water-stressed catchments, as it shows which crops require more irrigation water (blue water). In the work presented, it is used to demonstrate the *water deprivation* from the different sectors of the biosphere: green water use relates to the pedosphere (soil moisture), while blue water use relates to the hydrosphere (surface or groundwater). Due to data limitations, their distinction does not add value to the results presented. It can, though, be used for future reference for the further development of the methodology.
- In regards to the modelling and computations concerning the Urban Water Cycle, the implementation of a more detailed approach -especially in regards to the wastewater treatment plants- would exceed the scope of the research. The methodology created by Morera et al. (2016) was used as a basis for the more elaborate computations of the urban water footprint, while a number of assumption were made due to time and methodological constraints. Although the simplifications made affect the computed figures, they still enable to meet the original objective. That is, to show application of the Catchment Metabolism in practice and the use of water accounting methods as part of it.

Chapter 7: Discussion

The chapter discusses the character of the undertaken research and presents a critical analysis of the research outputs. It then analyses the methodological limitations of the work and the opportunities arising for further future research.

At first, the justification of the transdisciplinary nature of the research approach is presented. The critical analysis of the research outputs follows, presenting the strengths and limitations of the Catchment Metabolism modelling schema and of its application for integrated catchment analyses and asset management purposes. The relevance of the modelling approach to current regulatory challenges is then analysed, followed by a discussion on the areas for further research.

7.1. The transdisciplinary nature of the research

The principles of Transidsciplinarity, as defined by Leavy (2011) were introduced in Chapter 3. In this section, the principles underpinning the creation of the Catchment Metabolism modelling schema and the attributes of the schema per se are tested against the principles to reveal its transdisciplinary character (**Table 7.1.**).

Table 7.1.: The principles of Transidsciplinarity, according to Leavy (2011) and how the Catchment Metabolism modelling schema meets the criteria for being classified as a transdisciplinary research methodology.

Principle	Practice for Catchment Metabolism
Issue- or Problem- Centred (problem at the centre of research and guides methodology)	Research Problem: lack of standardised methodologies for integrating natural capital in the asset management portfolio of the water sector
Holistic or Synergetic Research Approach (problem considered holistically through an iterative process which produces integrated knowledge)	-Synergies created on the grounds of several scientific fields -Use of multiple diagrammatic representations of the catchment system to serve diverse research purposes
Transcendence (creation of conceptual frameworks that transcend discipline perspectives)	The modelling schema created is based on concepts and tools from different disciplines, ranging, for example, from Economics (Input-Output Analysis) to Hydrology (Water Inventories)
Emergence (emergence of new conceptual and methodological frameworks)	The Catchment Metabolism modelling schema has emerged as an output. Further, the combined use of multiple types of diagrams and water indices
Innovation (creation of new conceptual, methodological and theoretical frameworks)	The innovation of the work lies on the synthesis of concepts and methods for the creation of the modelling schema
Flexibility (iterative research process requiring adaptation to new insights)	'Retrospective' approach for the creation of the modelling schema. Its structure enables the integration of on-going advancements in the field of water inventories.

The motive for the creation of the Catchment Metabolism modelling schema was the limited research on structured methodologies for the integration of the natural capital in the asset management portfolio of the water industry. The creation of a holistic and structured modelling approach has been the focal point of the undertaken research (issue-centred) and has driven the methodological choices made throughout. A number of concepts and techniques were synthesised (synergetic research approach) for the creation of the modelling schema. The synergies were grounded on the principles of several scientific fields and their methods (transcendence), ranging from Integrated Catchment Management (ICM) and Hydrology (e.g. Water Budgets and Water Inventories), to Industrial Ecology (e.g. metabolism) and Economics (Input-Output Analysis). The 'retrospective' approach followed for the creation of the modelling schema shows evidence of a creative research process (flexibility), while the transparent structure of the schema enable its further development with the integration of the on-going advancements in the field of Water Inventories. The reproducible and scalable Catchment Metabolism modelling schema is the main output of the undertaken research (emergence). Its step-by-step creation process enables its adoption from the water industry to serve pragmatic challenges (issue-centred). Further, the use of multiple diagrammatic representations for catchment systems serves the principles of ICM and contributes to the creation of new knowledge (innovation) in the field.

The attributes presented and discussed qualify for the classification of the emergent methodology as transdisciplinary, after the definition of Leavy 2011. Further, following the definition of transidsciplinarity as a means to bring policy requirements into academic research (Stavridou and Ferreira 2010; Pohl 2008), the undertaken research qualifies as transdisciplinary for brining into the methodological choices the policy requirements of the UK water policy bodies (OFWAT, DEFRA) asking for methodologies which will ensure resilient water systems.

7.2. The Catchment Metabolism modelling schema: a critical analysis

The undertaken research introduces a structured approach for designing the Asset Management Plans (AMPs) of the water industry around a new focal point: the ecosystem. Built on the principles of Ecosystem Services, where the ecosystem is an equal stakeholder, the Catchment Metabolism modelling schema adds to the limited literature on the systemic approaches for the integration of natural capital in the asset management portfolio of the water sector. The schema introduced is designed on a robust, transdisciplinary basis but is also practical, so that it can be easily used from water practitioners. Its feasibility to serve everyday practice has been validated through its application for an industrial case study for a rural catchment system.

The work stresses the importance of assessing water-related issues and decision-making at a geographically bounded scale, that of the river basin or watershed or catchment. It therefore contributes to the discussion on the issue of the optimum scale for effective water resources management, as debated in recent literature (Stodden et al. 2016; Vörösmarty et al. 2015; Jaramillo and Destounni 2015). Despite the necessity for local analyses of water resources availability, the contribution of local stressors and impacts at a global level should not be neglected. On the contrary, they should be used as a key driver in the development of Earth system modelling. Counter wise, adopting and retaining a global perspective provides context to local conditions and enables crafting international stewardships for sustainable and equitable water management, especially in regards to global trade and transboundary water conflicts.

The Catchment Metabolism demonstrates a structured approach to achieve regional strategic planning which enables multiple perspectives in the analysis. Its coherent structure could inform the design of integrated catchment management strategies and assist the successful implementation of catchment-based initiatives. It introduces new patterns in conceptualising and modelling a catchment, collecting data and displaying information which allows for a better understanding of the sub-systems of complex systems. The creation and further development of systemic approaches at this scale would respond to the need for effective tools for supporting strategic decision-making and facilitating communication among stakeholders, ranging from water companies to regulatory bodies. Nonetheless, the scalability of the Catchment Metabolism and its underpinning methodologies provide a platform to explore and rigorously compare sustainable solutions at multiple scales, ranging from local and regional to national and global. The integration of indicators accounting for water stress and availability enable an analysis which considers the wider water resources context.

The systemic approach introduced is concise, scalable, flexible, re-producible and easy to use, as it is a step-by-step process. Although the focus of the research is the water cycle, the underpinning methodology of the modelling schema can be applied to other studies looking at the water, carbon or nitrogen natural cycles. In addition, the current work presents its application at a wide catchment (water basin) scale. However, it can be applied to diverse catchment systems, varying in size (from sub-catchments to tributaries) and metabolisms. The scope and scale of application may vary, but the underpinning rules applied and the steps undertaken would remain the same. Thus, for the reproduction of the approach for other catchment systems, the experts involved would need to follow the structured step-by-step procedure outlined in the results' chapters. The identification of the main actors of the catchment, their activities and interlinked relations would lead to the definition of its metabolism. The outputs for different catchment systems would vary dependant on the catchment's typology (natural setting and conditions) and metabolic compartments. The outputs would be further differentiated upon the performance of arithmetic calculations- based on catchment-specific investigations, water accounting techniques, data availability, and selection of water indices.

The clearly defined building blocks of the CM schema make it modular: parts of the methodology can be disseminated to experts and then assembled to formulate the modelling schema. The tools utilised to synthesise the methodology contribute to the delivery of a coherent approach and can all be reproducible from the actors involved in asset and catchment management projects. Based on the popular concept and methodology of environmental input-output analysis (E-IO) the Catchment Metabolism modelling schema opens the black box of natural flow accounting for business purposes. The Catchment PIOT captures the flows occurring both in the interface of biosphere and technosphere, but also within the biosphere alone. This attribute enables the use of the schema from a diverse audience of experts, ranging from hydrologists to asset managers and engineers and creates the ground for a shared format of information display.

The complexity of the endeavour of modelling aspects of the water cycles has been highlighted in literature (e.g. Valipour et al. 2015) although the value of the existing hydrological models for decision-making purposes is challenged (Haberlandt et al. 2009). The transparency of the Catchment Metabolism enables the detailed mapping of each of the subsystems of a catchment system and highlights the complexities of a catchment system which can be modelled and addressed by hydrological models. It therefore enables the integration of the outputs of existing hydrological models into policy and decision-making. It can also highlight areas where more robust models are required. It can also assist identifying data priorities, the optimum granularity level for data gathering, along with the most appropriate data formats for value adding activities, such as the improvement of available models. The structure and underpinning methodologies of the Catchment Metabolsm modelling schema responds to the urging need to face the lack of transparency and irreproducibility of hydrological modelling approaches and tools (Stodden et al. 2016).

There is an emerging consensus that accounting for environmental assets- including water resources - would provide a valuable, comprehensive and integrated information set to guide environmental management and monitoring and policy-making (Hein et al. 2015; Obst and Vardon 2014) in public and corporate levels. Likewise, as Richter (2003) suggests, the use of environmental flows research allows for a clearer explanation about the distinction between ecosystem functions and ecosystem services. Indeed, the methodology presented sheds light on this confusion: the function occurs as part of the stakeholder 'ecosystem' and the outputs of the function are either environmental flows –those that return to the environment- or ecosystem services – which are the 'economic flows' of the biosphere to the technosphere, therefore, the contribution of the environment to the human wellbeing. Making use of the literature on the economic valuation of ecosystem services, economic values and costs can be estimated for all quadrants of the Catchment PIOT. Therefore, it can serve as the ground to build an economic model. The supplementary use of Earth System Modelling (Arbault et al. 2014) would provide further details on how flows are circulated within the catchment boundaries, especially for those 'critical' flows for the environment, e.g. stock flows.

Further, recent works (Pedro-Monzonís et al. 2016a,b; Hein et al. 2015; Dimova et al. 2014; Obst and Vardon 2014; Čuček et al. 2012) have demonstrated that the environmental and water accounting approaches, although simple in nature, are resource intensive and require the collection of data from multiple stakeholders and the aggregation of information at different scales. The design and application of the Catchment Metabolism modelling schema suffers from the same issues while the dubious availability of the datasets and the aggregation of information in a uniform format increases its complexity. The transdisciplinary character of such works stress the need for knowledge exchange and alignment of perspectives. More example case study applications may provide further practical insights and facilitate the integration of the methodology in every day practice. Nevertheless, the introduction of functional modelling (through IDEF0) for data collection and information display facilitates these tasks and creates common ground for information display in a concise way. The inclusion of information regarding the controls and mechanisms of a system or a process allows for holistic views and approaches to be implemented.

The "footprint family" suite of indicators (Galli et al. 2012), the water footprint (WF), the ecological footprint (EF) and the carbon footprint (CF), have been applied in literature as the underpinning methods for multi-regional input-output (MRIO) analyses. Ewing et al. (2012) introduced the MRIO-F (Multi-Regional Input-Output-Footprint) model, as a method to calculate national and regional ecological and water footprint values at the product-specific level, utilising the generic MRIO framework. The coupling methodology introduced was aimed at the harmonisation and further improvement of the EF and WF computations and the linkage of MRIO with footprint databases for international trade and product supply-chains. The work suggested that the combination of the methods increased the transparency in the analysis and provided a structure for further methodological improvements. In this vein, the use of the WF suite of indicators has shaped the water inventory of the Catchment PIOT and the creation of the water accounts for the catchment's stakeholders. The WF indices were selected as a powerful communication tool (Vanham and Bidoglio 2013; Galli et al. 2012) whose arithmetic values could be integrated in a catchment-based hydrological analysis.

Nonetheless, the design of the methodologies underpinning the water accounts for the case study presented- the Poole Harbour Catchment- suffers from several limitations, some of which result from the WF methodology per se. For the computation of the WF components of the catchment's actors (e.g. water company, agriculture) and their assessment against local conditions, multiple data sources were used, selected to serve the research purposes within the limited timescales of the project. For instance, the computations of the crop-related WF components were based on hydrometric data from the CLIMAT (2.0, FAO 2012b) database, as catchment-specific data could not be integrated in the CROPWAT (8.0, FAO 2012a) software due to formatting issues. Similarly, several assumptions were made regarding the hydrological parameters of the example (Poole Harbour) and its adjacent catchments. Met Office (MORECS) data were not available at a catchment scale and thus, it was assumed that the hydrometric values from the two adjacent MORECS squares would represent the natural water budget of the example catchment. Same assumption applies for the crop types and their water requirements, which were computed based on data from meteorological stations (Exeter, Bournemouth) located in adjacent catchments. Data availability and consistency among databases are defined as the primary limitations of the WFA methodology (Vanhan and Bidoglio 2013). For the use of the methodology as part of the Catchment Metabolism modelling schema and its application across the UK water sector, data aggregated at a catchment scale are required. Aggregating existing data in the catchment level would facilitate and stimulate the industrial uptake of the schema.

Further, a number of assumptions and simplifications were made for the construction of the actors' water accounts. The simplifications for the agriculture-related WF computations align with assumptions made in previous works for other parts of the UK (Zhang et al. 2014, report no RESE000355). The WF methodology applied for the urban water cycle was based on a modified WF methodology as introduced in recent literature (e.g. Morera et al. 2016) and was populated with data provided from the project's industrial partner. Site-specific data were obtained for example infrastructure assets located in the Poole Harbour Catchment. The results are therefore only indicative and serve as a means to show the applicability of the methodologies created. In both cases, the accounts created suffer from the limitations of the grey water footprint methodology, which is not yet complete and requires further improvements and standardisation (Vanhan and Bidoglio 2013; Thaler et al. 2012).

Last but far from least, the WFA methodology is considered a rather partial tool, as acknowledged by the Water Footprint Network research group itself (Hoekstra et al. 2011). Thus, the water footprint indicators do not account for a number of hydrological aspects, such as flooding, nor for infinite resources closely linked to water resources, such as land. Thus, the use of the water footprint family as the underpinning method of the Catchment PIOT water accounts limits the environmental outputs computed to water-related alone. To overcome this limitation, the accounting or inventory of the Catchment PIOT could be re-designed based on models harmonising the footprint family indicators (e.g. Ewing et al. 2012). It could also be expanded to include processes occurring outside the catchment boundaries (e.g. the phase of fertiliser production), which have been treated as externalities for the undertaken research. The use of LCA as complementary tool for the environmental assessment of processes occurring outside of the system's (catchment) boundaries would enable a trans-boundary assessment. In such studies, caution should be paid in the possibility of double-counting water flows either between catchments or between processes, i.e. direct and indirect use of water. Uncertainty analysis methods could also be coupled with the modelling (e.g. Cai et al. 2016) to provide more robust approaches and results.

7.3. Opportunities arising: Future work

This section summarises the research opportunities arising from the work presented. These relate to the gaps of the current form of the Catchment Metabolism modelling schema or to its further development in order to contribute to other research fields and be transferred to industrial and policy practice.

The research undertaken to date has shown the application of the Catchment Metabolism schema for the creation of the Catchment Physical Input-Output Table (PIOT) describing physical, water-related flows and outputs. Nonetheless, it could be applied to studies related to other physical flows, such as energy. The construction of Catchment PIOTs for different flows would reveal trade-offs among strategic decisions. For example, it could show in a structured and straight-forward format whether wastewater purification increases energy consumption; thus, carbon emissions. The schema can also be used to compare the environmental performance of management scenarios for green (e.g. construction of wetlands) and grey (e.g. wastewater treatment plant) infrastructure solutions, responding to the current demand for ecosystem-based water resources management (Vörösmarty et al. 2015; Palmer et al. 2015).

The structure of the Catchment PIOT could be used to show monetary flows. These would result either from the monetisation or economic evaluation of the physical flows (e.g. water) or from cost or value-related computations. For the latter, the creation of a new methodology and the integration of relevant metrics and indexes would be necessary. Following the rationale of the creation of the methodologies underpinning the water-related Catchment PIOT, the research field of Life Cycle Costing (LCC) could serve as the basis of the methodology. A combination of literature from ecosystem services, cost and value modelling would then be employed for the creation of the methodology.

Further on the expansion of the Catchment Metabolism modelling schema, its water accounting and output methodologies could be coupled with Geographical Information Systems (GIS) to enable spatially-explicit models. The coupling of GIS with Life Cycle Assessment (LCA) is gaining popularity in literature (e.g. Geyer et al. 2010) for both inventory and impact assessment modelling. These studies mainly look at the impacts of land use on biodiversity. The coupling of the Catchment Metabolism and its metrics with GIS would require the coupling of water footprinting techniques with GIS programming languages (e.g. Python) in order to support the creation of water-related thematic maps. These would consist of water accounting figures, such as water footprints, or of water-related outputs, such as water pollution levels. The creation of such maps would inform decision-making and the communication of the research results to non-expert audiences.

The combination of Life Cycle Thinking and Integrated Catchment Management has formulated the rationale of the Catchment Metabolism schema and arguments for the implementation of holistic, synergistic asset management. Future work may further investigate the synergies between the two research fields and, more specifically, explore the application of consequential Life Cycle Assessment (c-LCA) for catchment-based assessments. The creation of novel approaches for spatially and temporally explicit LCAs would respond to the current methodological challenges of LCA, requiring uniform systems boundaries and the integration of the temporal and dynamic components in the LCA methodology (McManus and Taylor 2015). Such methodological improvements could not only contribute to the standardisation of water-related c-LCA for regional scale indicators, characterisation factors and environmental analyses, but also prove beneficial for LCA applications, such as, land use and bioenergy.

For the work presented, the Catchment Physical Input-Output Table (Catchment PIOT) is destined for single-region assessments, as the outputs depicted in the table are deriving from the stakeholders/sectors located in a particular region: the catchment. The same principles would apply for studies looking at transboundary water issues (between multiple catchments) where multi-regional input output analysis would be applicable. Further, the case study presented refers to a rural catchment, whose metabolism has been analysed for proof-of-concept purposes. Nonetheless, each catchment is unique and needs to sustain a certain metabolism on which its whole internal structure depends. Thus, in order to sustain the particular metabolism of each catchment or catchment 'type', their thorough, systemic and holistic study is required. Further research on the generalisation of the Catchment Metabolism schema for typologies of catchments would involve the application of the schema for multiple, diverse catchments and metabolisms. The classification of catchments according to the Water Framework Directive (2000/60/EC) based on the challenges faced by the water courses or catchment types according to water uses and stakeholders would serve as future research case projects. The further development of the Catchment Metabolism schema would require the active involvement and input from different experts, which would serve the creation of knowledge blocks and ensure the quality of the data displayed and produced. The automatisation of some of the processes involved in the creation of Catchment PIOTs would facilitate the re-production of the schema for multiple catchments systems. A sophisticated Systems Dynamics (SD) model would be required, so that the mathematical modelling underpinning the creation of the stakeholders' Water Accounts (chapter 6) is embedded in the catchment SD mapping.

The creation of a sophisticated SD model would then inform catchment-based analyses in research areas relating to the nexus of water-food-energy (e.g. UN World Water Development Report, 2014) and to the co-ordination among relevant policies (Gleick 2016). Expanding the use of the schema is the nexus-related research, would bring the principles of Integrated Catchment Management into the areas of Ecosystem Services and Nexus Mapping. Literature (e.g. Malinga et al. 2015; Grêt-Regamey et al. 2014) shows a trending interest towards the synthesis of these fields and the creation of novel, transdisciplinary methodologies to support sustainable resource management.

The Catchment Metabolism schema has been formulated as an approach to integrate and account for other forms of capital, such as the natural capital, in the asset management portfolio of the water industry. It is a policy-driven modelling schema, which is designed in order to improve the transparency in the environmental accounting of the Asset Management Programmes (AMPs) of the water sector and facilitate the communication with the regulators. It is underpinned by clearly-defined rules and its creation is described as a logical, step-by-step process. The schema provides structure which enables the further expansion and development of the underpinning methodology, in order to serve future industry and policy demands. For example, the Catchment PIOT can be used as canvas for embedding sophisticated hydrological models describing either natural phenomena or processes among catchment's actors. The rationale and the unpinning rules and methods of the Catchment Metabolism schema and its inventories could also be used to formulate a structured approach for integrating the social capital in the strategic planning of the water industry. This expansion of the schema would be explored in the context of the field of Socio-hydrology, i.e. the field which explores the integrated human-hydrology systems and the co-evolving dynamics, feedbacks and behaviours across multiple time and space scales (Blair and Buytaert 2016; Elshafei et al. 2014). Future research in this vein would also involve the exploration of the influence of catchment-based, integrated and holistic asset management planning on the tariff policy of the UK water sector.

Chapter 8: Conclusions

The chapter summarises the research tasks undertaken and maps them against their research outputs. Then, the original contribution of the research project is discussed, followed by the concluding remarks.

8.1. Summary of Research Tasks and Outputs

A number of research tasks were undertaken in order to meet objectives of the research project undertaken. The research outputs, as mapped against the project's objectives, are summarised below (**Table 8.1.**).

- **i.** The rationale of '*Holistic Asset Management*' (section 5.1.) involves the development of strategies which enable the integration of several forms of capital (e.g. built, natural) in the asset management portfolio of the water industry. Further, it stresses the need for tackling catchment-specific issues, such as pollution, in synergistic approaches.
- **ii.** The Catchment Metabolism modelling schema (sections 5.2.; 5.3.) is grounded on Life Cycle Thinking and created based on the synthesis of techniques from the field of Industrial Ecology. The conceptualisation of the catchment systems has been formulated by the use of multiple diagrammatic representations. All the techniques used were selected for their ability to serve the research goal.
- **iii.** The environmental assessment of the performance of holistic asset management strategies was based on the creation of Water Inventories (chapter 6) for each of the stakeholders identified for a given catchment system (section 5.3.; chapter 6). The methods selected for the creation of the Water Inventories are grounded on the fields of hydrology and water accounting and enable the rigorous analysis of the water regimes of multiple water actors, such as the ecosystem, a water company or the sector of agriculture.
- **iv.** The applicability of the modelling schema, water inventories and their underpinning methodologies was tested through a case study: the Poole Harbour Catchment (chapter 4; section 6.5.). A number of scenarios were formulated, each representing a strategy for the synergistic tackle of nitrogen pollution in the Poole Harbour Catchment. A Catchment Physical Input-Output Table was created for each of the strategies, enabling the analysis and comparison of their environmental performance.
- **v.** The evaluation of the research outcomes and their practical value were discussed in a critical analysis (section 7.2.). The methodological strengths and limitations of the of the Catchment

Metabolism modelling schema were identified, followed by a discussion on opportunities for academic and industrial purposes (section 7.3.).

Table 8.1.: Research outputs mapped against research objectives.

Research Objectives	Research outputs
(i) Define '*Holistic Asset Management*'	-Integration of several forms of capital (e.g. built, natural) in the asset management portfolio -Synergistic approaches for tackling issues.
(ii) Select techniques & Define rules for the creation of the catchment-based modelling schema	-Modelling schema grounded on Life Cycle Thinking and techniques from the field of Industrial Ecology. -Techniques selected for serving the research goal.
(iii) Determine the tools & create the rules for the assessment of the environmental performance of holistic asset management strategies	-Stakeholders analysis and Water Inventories. -Methods selected to enable a scientifically rigorous analysis.
(iv) Investigate the applicability of the research outcomes through an industrial case study	Case study (Poole Harbour Catchment) and Scenario Analysis (Chapter 4, section 6.5.).
(v) Evaluate the practical value of the research outcomes	Critical Analysis (section 7.2.).

8.2. Contribution to Knowledge

The research contributes a transdisciplinary, whole-systems approach for conceptualising, modelling and analysing catchments as complex asset systems.

It provides a unique, novel, comprehensive and structured methodology which allows the integration of natural capital in the asset management portfolio of the water industry. Its creation is based on the synthesis of concepts, tools and methods from a spectrum of disciplines.

The modelling methodology includes two main features: a modelling schema and a modelling inventory. The rules underpinning the creation of both the schema and the inventory are the product of a robust knowledge assembly.

The novelty created from this research encompasses the underpinning methodological approach through to the specific rules for analysing the catchment. **Figure 8.1.** depicts the contribution headings, with the rules being the core contribution to knowledge.

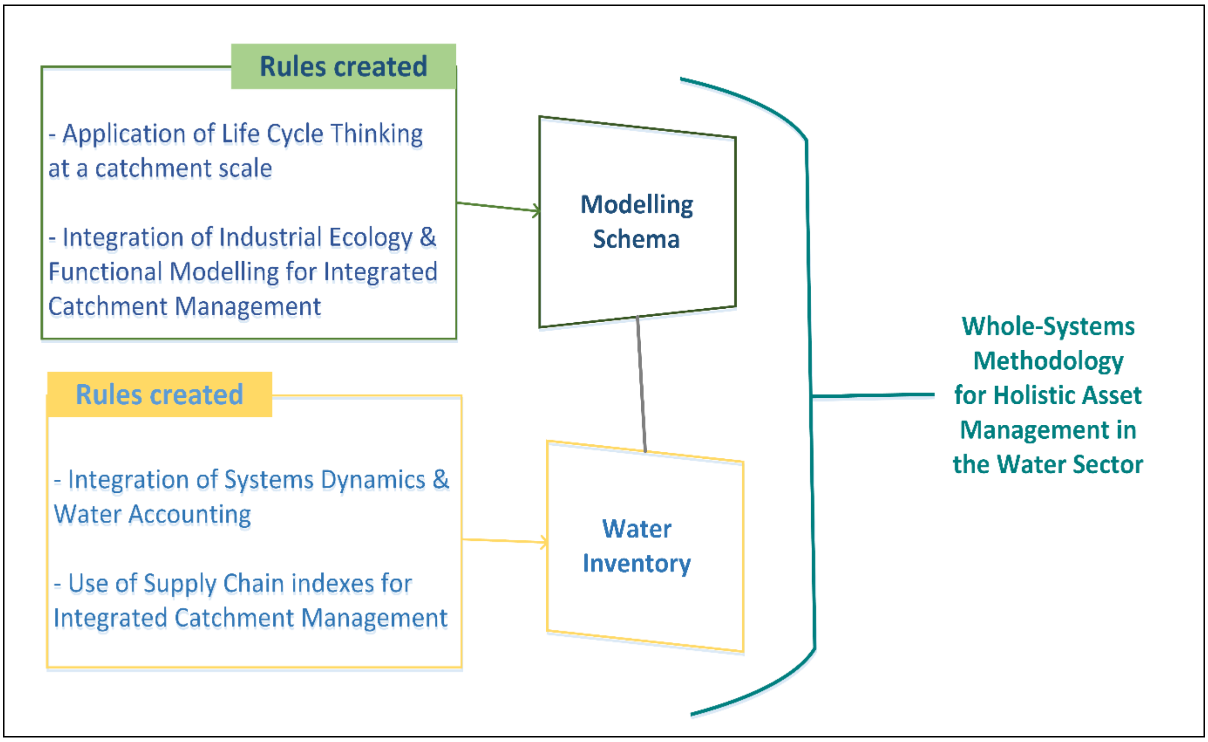


Figure 8.1.: Research's Contribution to Knowledge.

In detail:

- The methodological approach has determined the boundaries and context to enable a catchment (watershed) to be defined as a unit of analysis within asset management.
- Using systems thinking, the catchment has been represented as an integrated system with inputs, outputs and outcomes of the system being identified, enabling a transparent approach for holistic asset management.
- The research has defined the rules for the application of life cycle thinking at a catchment scale through the integration of tools from the field of industrial ecology and functional modelling. The schema created has been based on these rules and enables the implementation of holistic asset management at a catchment scale.
- The research determined the rules for the integration and joint use of Systems Dynamics and Water Accounting. This integration has informed the creation of a whole-systems water inventory for flow accounting at a catchment scale, whose outputs feed into the modelling schema. It has also investigated the use of indexes from supply chain and product systems management in Integrated Catchment Management.

Together, the methodological approaches presented in this research have integrated concepts and techniques on a transdisciplinary basis and thus, enable cross-disciplinary knowledge transfer and transidsciplinarity in practice. They have investigated synergies among disciplines and research fields, have demonstrated a sophisticated knowledge synthesis process and have created a transparent and re-applicable methodology. The research expands the scope of asset management of the water sector, as it assists in addressing challenges related to regulatory compliance and strategic planning.

8.3. Epilogue

To the author's view, the research undertaken is a water-related fugue, whose theme is defined as 'sustainable water use'. A structured and meticulous knowledge assembly has enabled the delivery of the research outputs, which are grounded on multiple fields. Each of these fields -or voices of the fugue- contributed to the creation of robust methodologies for analysing complex systems. The multiple 'voice' combinations created have resulted from an iterative research process, as inspired by the study process of a musical fugue: after the definition of the theme and the thorough study of the voices (research fields), synergies were identified and connections were made, aiming to the creation of a 'tuned' outcome, that preserved the unique features of its components.

The fugue metaphor has enabled the implementation of creativity in practice and the design and delivery of an original, truly transdisciplinary research project.

List of References

AASB and BoM (Australian Accounting Standard Board and Bureau of Meteorology) (2012). Australian Water Accounting Standard 2: Assurance Engagement on General Purpose Water Accounting Reports. Commonwealth of Australia, Melbourne.

Allan, J.A. (2003). Virtual water-the water, food and trade nexus: Useful concept or misleading metaphor? Water International 28(1), 106-113.

Allan, A. (2012). Is there a role for water accounting in the avoidance and resolution of international water disputes? In: Godfrey, J., Chalmers, K. (Eds.), International Water Accounting: Effective Management of a Scarce Resource. Edward Elgar Publishing Inc., New York, pp. 236-254.

Alberts, E.E., Schuman, G.E., Burwell, R.E. (1978). Seasonal runoff losses of nitrogen and phosphorus from Missouri Valley loess watersheds. Journal of Environmental Quality 7(2), 203-208.

Allby, M. (2008). Oxford Dictionary of Earth Science, 3rd Edition, Oxford University Press, ISSN 978-0-19-921194-4.

Angrill, S., Farreny, R., Gasol, C.M., Gabarrell, X., Viñolas, B., Josa, A., Rieradevall, J. (2012). Environmental analysis of rainwater harvesting infrastructures in diffuse and compact urban models of Mediterranean climate. International Journal of Life Cycle Assessment (2012) 17:25-42.

Arbault, D., Rivière, M., Rugani, B., Benetto, E., Tiruta-Barna, L. (2014). Integrated earth system dynamic modelling for life cycle impact assessment of ecosystem services. Science of the Total Environment 472, 262-272.

Arthington, A.H. (2012). Environmental Flow: Saving Rivers in the Third Millennium. University of California Press, Berkeley and Los Angeles, California, USA.

Avila-Foucat, V.S., Perrings, C., Raffaelli, D. (2009). An ecological-economic model for catchment management: The case of Tonameca, Oaxaca, Mexico. Ecological Economics 68, pp. 2224-2231.

Balali, H., Viaggi, D. (2015). Applying a System Dynamics approach for modelling groundwater dynamics to depletion under different economical and climate change scenarios. Water 7, pp. 5258-5271.

Barjoveanu, G., Comandaru., I.M., Rodriguez-Garcia, G., Hospido, A., Teodosiu, C. (2014). Evaluation of water services system through LCA. A case study of Iasi City, Romania. International Journal of Life Cycle Assessment 19, pp. 449-462.

Barrios, R., Siebel, M., van der Helm, A., Bosklopper, K., Gijzen, H. (2008). Environmental and financial life cycle impact assessment of drinking water production in Waternet. Journal of Cleaner Production 16, pp. 471-476.

Basset-Mens, C., Anibar, L., Durand, P., van der Wrf, H.M.G. (2006). Specialised fate factors for nitrate catchments: Modelling approach and implication for LCA results.

Baumann. H., Tillman, A.M., 2004. The Hitch Hiker's guide to LCA. An orientation in life cycle assessment methodology and application. Studentlitteratur. ISBN 978-44-02364-9.

Bayart, J., Bulle, C., Deschenes, L., Margni, M., Pfister, S., Vince, F., & Koehler, A. (2010). A framework for assessing off-stream freshwater use in LCA. International Journal of Life Cycle Assessment, 15(5), 439-453. doi: 10.1007/s11367-010-0172-7.

Bayart, J., J. Grimaud and E. Aoustin (Submitted). The Water Impact Index: a simplified single-indicator approach for water footprinting." *The International Journal of Life Cycle Assessment*. Accessed on: http://www.wulca-waterlca.org/publications.html.

Bello-Maurel, V., Short, M.D., Roux, P., Schulz, M., Peters, G.M. (2014). Streamlining life cycle inventory data generation in agriculture using traceability data and information and communication technologies-concepts and technical basis. Journal of Cleaner Production 69, pp. 60-66.

Berger, M., Pfister, S., Motoshita, M. (2016). Water Footprinting in Life Cycle Assessment: How to count the drops and assess the impacts? LCA Compendium – The Complete World of Life Cycle Assessment. Special Types of Life Cycle Assessment, pp.73-114.

Berger, M., van der Ent, R., Eisner, S., Bach, V., Finkbeiner, M. (2014). Water accounting and vulnerability evaluation (WAVE)- considering atmospheric evaporation recycling and the risk of freshwater depletion in water footprinting. Environmental Science and Technology 48(8):4521-4528.

Berger, M., Finkbeiner, M. (2010). Water Footprinting: How to address water use in Life Cycle Assessment? Sustainability 2, pp. 919-944.

Bernard, J.H., Zhou, J., Giannis, A., Chang, V.W.C., Want, J.Y. (2014). Environmental life cycle assessment of different domestic wastewater streams: Policy effectiveness in a tropical urban environment. Journal of Environmental Management 140, pp/ 60-68.

Billen, Gilles (2003). From Ecology of Natural Systems to Industrial Ecology. Eds: Dominique Bourg and Suren Erkman. Greenleaf Publishing Limited, UK.

Billen, G., Silvestre, M., Grizzetti, B., Leip, A., Garnier, J., Voss, M., Howarth, R., Bouraoui, F., Lepistö, A., Kortelainen, P., Johnes, P., Curtis, C., Humborg, C., Smedberg, E., Kast, Ø., Ganeshram, G., Beusen, A., Lancelot, C. (2011). Nitrogen flows from European regional watersheds to coastal marine waters. In: The European Nitrogen Assessment. Sutton, M.A. et al (Eds), Cambridge University Press 2011.

Blair, P. and Buytaert, W. (2016). Socio-hydrological modelling: a review asking "why, what and how?". Hydrology and Earth System Sciences 20, pp. 113-178.

Blizzard, J. L.; Klotz, L. E. (2012). A framework for sustainable whole systems design. Design Studies 33 (5), pp. 456–479.

BoM (Bureau of Meteorology) and Water Accounting Standard Board (2014). Water Accounting Conceptual Framework for the Preparation and Presentation of General Purpose Water Accounting Reports. Commonwealth of Australia, Melbourne.

Bonton, A., Bouchard, C., Barbeau, B., Zedrzejak, S. (2012). Comparative life cycle assessment of water treatment plants. Desalination 284, pp. 42-54.

Boulay, A., Bouchard, C., Bulle, C., Deschenes, L., & Margni, M. (2011). Categorising water for LCA inventory. International Journal of Life Cycle Assessment., 16(7), 639-651. doi: 10.1007/s11367-011-0300-z.

Boulay, A.M., Hoekstra, A.Y., Vionnet, S. (2013). Complementarities of Water-Focused Life Cycle Assessment and Water Footprint Assessment. *Environmental Science and Technology* 47:11926-11927.

Boulay, AM., Bare, J., De Camillis, C., Döll, P., Gassert, F., Gerten, D., Humbert, S., Inaba, A., Itsubo, N., Lemoine, Y., Margni, M., Motoshita, M., Núñez, M., Pastor, A. V., Ridoutt, B., Schencker, U., Shirakawa, N., Vionnet, S., Worbe, S., Yoshikawa, A. and Pfister, S. (2015a). Consensus Building on the development of a stress-based indicator for LCA-based impact assessment of water consumption: Outcome of the expert workshops. The International Journal of Life Cycle Assessment 20.5 (2015): 577-583.

Boulay, AM., Bayart, J., Bulle, C., Franceschini, H., Motoshita, M., Muñoz, I., Pfister, S. and Margni, M. (2015b). Analysis of Water Use Impact Assessment Methods (Part B): Applicability for Water Footprinting and decision making with a laundry case study. *The International Journal of Life Cycle Assessment* 20.6 (2015): 865-879.

Boulay, AM., Motoshita, M., Pfister, S., Bulle, C., Muñoz, I., Franceschini, H. AND Margni, M. (2015c). Analysis of Water Use Impact Assessment Methods (Part A): Evaluation of modelling choices based on a quantitative comparison of scarcity and human health indicators. The International Journal of Life Cycle Assessment 20.1 (2014): 139-160.

Boulay, AM. et al. (2016). The AWARE method. Submitted. http://wulca-waterlca.org/project.html

Bracken, L.J., Bulkeley, H.A., Whitman, G. (2015). Transdisciplinary research: understanding the stakeholder perspective. Journal of Environmental Planning and Management 28(7), pp. 1291-1308.

Brandão, M., Clift, R., Mila I Canals, l., Basson, L. (2010). A life-cycle approach to characterising environmental and economic impacts of multifunctional land-use systems: an integrated assessment in the UK. Sustainability 2, pp. 3747-3776.

Brattebø, Helge (2003). The impact of Industrial Ecology on University Curricula. *Perspectives on Industrial Ecology*. Eds: Dominique Bourg and Suren Erkman. Greenleaf Publishing Limited, UK.

Brisbane Declaration (2007). The Brisbane Declaration: Environmental Flows are Essential for Freshwater Ecosystem Health and Human Well-Being, September3–6, Brisbane, Australia.

Brown, J.H., Gillooly, J.F., Allen, A.P., Savage, V.M., West, G.B. (2004). Toward a metabolic theory of ecology. Ecology 85, pp. 1771-1789.

Brunner, P.H. and Rechburger, H. (2003). Practical Handbook of Material Flow Analysis. Lewis Publishers, CRC Press Company.

BS ISO/DIS 14040 (2006). International Standard. Environmental Management - Life Cycle Assessment-Principles and framework. International Organisation for Standardization, Geneva.

BS ISO/DIS 14044 (2006). International Standard. Environmental Management- Life Cycle Assessment-Requirements and Guidelines. International Organisation for Standardization, Geneva.

BS ISO/ DIS 14046 (2014). Water footprint-Principles, requirements and guidelines. International Organisation for Standardization, Geneva.

BSI, British Standards Institutions, 2008. Public Available Specification PAS 55:1&2: Asset Management. London. ISBN 978-0-580-50976-6.

BSI, British Standards Institutions, 2008. BS ISO 15686-5:2008. Buildings and constructed assets – Service life planning- Part 5: Life cycle costing.

Butler, J.R.A., Wong, G.Y., Metcalfe, D.J., Honzák, M., Pert, P.L., Rao, N., van Grieken, M.E., Kawson, T., Bruce, C., Krron, F.J., Brodie, J.E. (2011). An analysis of trade-offs between multiple ecosystem services and stakeholders linked to land use and water quality management in the Great Barrier Reef, Australia. Agriculture, Ecosystems and Environment, pp. 1-16.

Caffrey, K.R., Veal, M.W. (2013). Conducting and agricultural life cycle assessment: Challenges and Perspectives. The Scientific World Journal, pp.1-13.

Cai, Y., Yue, W., Xu, L., Yang, Z., Rong, Q. (2016). Sustainable urban water resources management considering life cycle environmental impacts of water utilisation under uncertainty. Resources, Conservation and Recycling 108, pp. 21-40.

Cammack, R. (Ed) (2006). Oxford Dictionary of Biochemistry and Molecular Biology, 2nd edition, Oxford University Press, ISBN 0-19-852917-1.

Cameron, F., Mengler, S. (2009). Complexity, Transidsciplinarity and Museum Collections Documentation. Emergent Metaphors for a Complex World. *Journal of Material Culture* Vol. 14(2): 189-218.

Chapagain, A.K., Hoekstra, A.Y., Savenije, H. H.G., Gautam, R. (2006). The water footprint of cotton consumption: an assessment of the impact of worldwide consumption of cotton products on the water resources in the cotton producing counties. *Ecological Economics* 60(1), pp. 186-203.

Chenoweth, J., Hadjikakou, M., Zoumides, C. (2014). Quantifying the human impact on water resources: a critical review of the water footprint concept. *Hydrology and Earth System Sciences* 18: 325-2342. 10.5194/hess-18-2325-2014.

Ciroth, A., Huppes, G., Klöpffer, W., Rüdenauer, I., Steen, B., Swarr, T. (2008). Environmental Life Cycle Costing. Hunkeler, D., Lichtenvort, K., Rebitzer, G. (Eds). CRC Press, Taylor & Francis Group. Published in collaboration with SETAC. p.184. ISBN 978-1-4200-5470-5.

Clift, R., Druckmna, A., Christic, I., Kennedy, C., Keirstead, J. (2015). Urban metabolism: a review in the UK context. Government Office for Science, No GS/15/30, London, UK.

Cook, B.R. and Spray, C.J. (2012). Ecosystem services and integrated water resources management: Different paths to the same end? Journal of Environmental Management 109, pp. 93-100.

Corominas, Ll., Foley, J., Guest, J.S., Hospido, A., Larsen, H.F., Morera, S., Shaw, A. (2013). Life cycle assessment applied to wastewater treatment: State of the art. Water Research 47, pp. 5480-5492.

Čuček, L., Klemes, J.J., Kravanja, Z. (2012). A review of footprint analysis tools for monitoring impacts on sustainability. Journal of Cleaner Production 34 (2012): 9-20.

Curran, M.A., Mann, M., Norris, G. (2005). The international workshop on electricity data for life cycle inventories. Journal of Cleaner Production 13 (8), 853–862.

Daniels, P.L., Lenzen, M., Kenway, S.J. (2011). The ins and outs of water use- a review of multi-region input-output analysis and water footprints for regional sustainability analysis and policy. Economic Systems Research, 23:4, 353-370.

De Baan, L., Alkemade, R., Koelner, T. (2013). Land use impacts on biodiversity in LCA: a global approach. International Journal of Life Cycle Assessment 18, pp. 1216-1230.

De Marco, O., Lagioia, G., Amicarelli, V., Sgaramella, A. (2009). Constructing Physical Input-Output Tables with Material Flow Analysis (MFA) Data: Bottom-up case studies. *S. Suh (ed.), Handbook of Input-Output Economics in Industrial Ecology*, Eco-Efficiency in Industry and Science 23, Springer.

De Souza, D.M., Flynn, D.F.B., DeClerk, F., Rosenbaum, K., de Melo Lisboa, H., Koelner, T. (2013). Land use impacts on biodiversity in LCA: proposal of characterisation factors based on functional diversity. International Journal of Life Cycle Assessment (2013) 18:1231-1242.

Den Hond, F. (2000). Industrial Ecology: a review. Regional Environmental Change 1(2), pp. 60-69.

Denscombe, Martyn (2010). The Good Research Guide: for small-scale social research projects (4th edition). Open University Press.

Department for Environment Food & Rural Affairs, DEFRA (2016). Creating a great place for living: Enabling resilience in the water sector. Report PB14418. March2016, London, UK.

Department for Environment Food & Rural Affairs, DEFRA (2011). An Inventory of Mitigation Methods and Guide to their Effects on Diffuse Water Pollution, Greenhouse Gas Emissions and Ammonia Emissions from Agriculture. Project WQ106.

Department for Environment Food & Rural Affairs, DEFRA (2005). Agricultural census data for selected catchments.

Dharmaratna, D., Harris, E. (2012). Estimating Residential Water Demand Using the Stone-Geary Functional Form: The Case of Sri Lanka. *Water Resources Management* 26 (8), 2283-2299; 10.1007/s22269-012-0017-1.

Dimova, G., Tzanov, E., Ninov, P., Ribakova, I. Kossida, M. (2014). Complementary use of the WEAP model to underpin the development of SEEAW physical Water use and supply tables. In: 12th International Conference on Computing and Control for the Water Industry, CCMI2013. *Procedia Engineering* 70, 563-572.; 10.1016/j.proeng.2014.02.062.

Dingman, S.L. (1992). Physical Hydrology. Prentice Hall, Upper Savage, New Jersey.

Echelai, G.A. (2013). Development of an asset management framework: the case study of NWSC, Uganda. Water Asset Management International, Issue 3, Vol.9, 11-15.

Earles, J.M., Halog, A. (2011). Consequential life cycle assessment: a review. International Journal of Life Cycle Assessment 16, pp. 445-453.

Ehrenfeld, John (2004). Industrial Ecology: a new field or only a metaphor? Journal of Cleaner Production 12 (2004), 825-831, doi:10.1016/j.jclepro.2004.02.003.

Elshafei, Y., Coletti, J.Z., Sivapalan, M., Hipsey, M. (2015). A model of the socio-hydrologic dynamics in a semiarid catchment: isolating feedbacks in the coupled human-hydrology system. Water Resources Research 51, 6442-6471.

Eriksson, O., Bisaillon, M., Haraldsson, M., Sundberg, J. (2014). Integrated waste management as a mean to promote renewable energy. Renewable Energy 61, pp. 38-42.

Erkman, Suren. (2003). Perspectives on Industrial Ecology. Eds: Dominique Bourg and Suren Erkman. Greenleaf Publishing Limited, UK.

Environment Agency, Natural England, Wessex Water Services Ltd (2013). Strategy for managing nitrogen in Poole Harbour Catchment to 2015. Report.

European Commission (2000) 'Directive 2000/60/EC of the European Parliament of the Council of 24 23rd October 2000 establishing a framework for community action in the field of water policy', Official Journal of the European Communities.

European Commission (2012). 'A Water Blueprint for Europe'. Luxembourg: Publications Office of the European Union. ISBN 978-92-79-30543-6.

Eurostat. (2001). Economy-wide material flow accounts and derived indicators. A methodological guide. European Commission. Luxemburg. ISN 92-894-0459-0.

Ewing, B.R., Hawkins, T.R., Wiedmann, T.O., Galli, A., Ercin, A.E., Weinzettel, J., Steen-Olsen, K. (2012). Integrating ecological and water footprint accounting in a multi-regional input-output framework. Ecological Indicators 23, pp. 1-8.

Fandel, C. (2012). Hydrogeology. Objectives, Methods, Applications. Science Publishers, pp. 12-14. 10.1201/b12932-5.

FAO (Food and Agriculture Organisation) (2012a). 'CLIMWAT 2.0 database'. http://www.fao.org/nr/water/infores_databases_climwat.html.

FAO (Food and Agriculture Organisation) (2012b). 'CROWAT 8.0 model'. http://www.fao.org/nr/water/infores_databases_cropwat.html.

Farreny, R., Rieradevall, J., Barbassa, A.P., Teixeira, B., Gabarrell, X. (2013). Indicators for commercial urban water management: the case of retail parks in Spain and Brazil. *Urban Water Journal*, 10 (4), 281-290.

Farreny, R., Oliver-Solà, J., Escuder-Bonilla, S., Roca-Marti, M., Sevigné, E., Gabarrell, X., Rieradevall, J. (2012). The metabolism of cultural services. Energy and flows in museums. *Energy and Buildings* 47: 98-106.

Federation of Canadian Municipalities (FCM) and National Research Council (2005). Managing Infrastructure Assets.

Feldmann, C.G. (1998): The Practical Guide to Business Process Reengineering Using IDEF0. Dorset House Publishing, 218p., ISBN: 0-932633-37-4

Feng, K., Siu, Y.L., Guan, D., Hubacek, K. (2011). Assessing regional virtual water flows and water footprints in the Yellow River Basin, China: A consumption based approach. *Applied Geography* 32(2011): 691-701.

Finnveden, G., Hauschild, M.Z., Ekvall, T., Guineé, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., Sangwon, S. (2009). Recent developments in Life Cycle Assessment. Journal of Environmental Management 91, pp. 1-21.

FIPS PUB (1983): Integration Definition for Function Modelling (IDEF0), U.S. Department of Commerce, Technology, Administration, National Institute of Standards and Technology (Washington, D.C.: 1983).

Fischer-Kowalski, Marina (2003). On the history of Industrial Metabolism. Eds: Dominique Bourg and Suren Erkman. Greenleaf Publishing Limited, UK.

Flood, R.L. (1999). Rethinking the fifth discipline: Learning within the unknowable. London, Routledge.

Friedrich, E., Pillay, S., Buckley, C. (2007). The use of LCA in the water industry and the case of an environmental performance indicator. Water SA 33 (4), pp. 443-452.

Galli, A., Wiedmann, T., Ercin, E., Knoblauch, D., Ewing, B., Giljum, S. (2012). Integrating ecological, carbon and water footpring into a footprint family of indicators: definition and role in tracking human pressure on the planet. Ecological Economics 16, pp. 100-112.

Gan, H., Wang, Y., Lu, Q., Vardon, M., Chanchai, A. (2012). Development and Application of the System of Environmental-Economic Accounting for Water in China. In: Godfrey, J., Chalmers, K. (Eds). International Water Accounting of a Scarce Resources. Edward Elgar Publishing Inc., New York, pp. 139-161.

Gerbens-Leenes, P.W., Mekonnen, M.M., Hoekstra, A.Y. (2013). The water footprint of poultry, pork and beef: a comparative study in different countries and production systems. Water Resources and Industry 1-2 (2013):25-36.

Geyer, R., Stoms, D.M., Lindner, J.P., Davis, F.W., Wittstock, B. (2010). Coupling GIS and LCA for biodiversity assessment of land use. International Journal of Life Cycle Assessment (2010) 15:454-467.

Giljum, Stefan and Hubacek, Klaus (2009). Conceptual foundations and Applications of Input-Output Tables. Handbook of Input-Output Economics in Industrial Ecology. Ed.: Sangwon Suh. Springer.

Giri, A.K., 2002. The Calling of a Creative Transdisciplinarity. Futures, 34(1), 103-115.

Gleeson, T., Wada, Y., Bierkens, M.F.P., van de Beek, L.P.H. (2012). Water balance of global aquifers revealed by groundwater footprint. Nature Letters, Vol. 488, pp. 197-200, doi:10.1038/nature11295.

Gleick, P.H. (2016). Water strategies for the next administration. Science 354 (6312), pp. 555-556.

Gooday, R., Hockridge, B., Lee, D., Stockely, V., Wright, E. (2015). Solent Harbours Nitrogen Management Investigation. Report from ADAS UK Ltd.

Gooday, R., Anthony, S.G., Chadwick, D.R., Newell-Price, P., Harris, D., Duethmann, D., Fish, R., Collines, A.L., Winter, M. (2014). Modelling cost-effectiveness of mitigation methods for multiple pollutants at farm scale. Science of the Total Environment 468-469, pp.1198-1209.

Gooday, R., Anthony, S. (2010). Mitigation Method-Centric Framework for Evaluating Cost-Effectiveness. Report from ADAS UK Ltd.

Godskesen, B., Hauschild, M., Rygaard, M., Zambrano, K., Albrechtsen, H-J. (2013). Life-cycle and freshwater withdrawal impact assessment of water supply technologies. Water Resources 47 (2013):2363-2374.

Goedkoop M., Heijungs R., Huijbregts M., de Schryver A., Struijs J., van Zelm, R. (2009). ReCiPe 2008: A life cycle assessment method which comprises harmonized category indicators at the midpoint and the endpoint level. Report I: characterisation. Ministry of Housing, Spatial Planning and Environment, Amsterdam.

Grêt-Regamey, A., Weibel, B., Kienast, F., Rabe, S-E., Zulian, G. (2014). A tiered approach for mapping ecosystem services. Ecosystem Services 13, pp. 16-27.

Guinée, J.B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H.A., de Bruijn, J.A., van Duin, R., Huijbregts, M.A.J. (2002). Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards. Series: Eco-efficiency in Industry and Science. Kluwer Academic Publishers, Dordrecht.

Guinée, J.B., Heijungs, R., Huppes, G. (2011). Life Cycle Assessment: Past, Present, Future. Environmental Science and Technology (2011), vol.45, no.1. doi:10.1021/es101316v

Haberlandt, U., Buchwald, I., van der Hijden, S., Verworn, A. (2009). Requirements of hydrological models to be used as part of decision-support systems in integrated water resources management. Hydrological Sciences Journal (IAHS-AISH Publication), Vol. 327, 2009, 29-35.

Hainsworth, G., Salvi, G. (2014). Great Britain Water Industry for Dummies. A Wiley Brand. John Wiley & Sons Ltd. Chichester, England.

Hardie, M.A., Cotching, W.E., Doyle, R.B., Lisson, S. (2012). Influence of climate, water content and leaching on seasonal variations in potential water repellence. Hydrological Processes 26, 2041-2048.

Hawkins, T., Hendrickson, C., Higgins, C., Matthews, S. (2007). A mixed input-output model for environmental life cycle assessment and material flow analysis. Environmental Science and Technology 41, 1024-1031.
Heijungs, R., Settanni, E., Guinée, J. (2013). Toward a computational structure for life cycle sustainability analysis: unifying LCA and LCC. International Journal of Life Cycle Assessment 18, pp. 1722-1733.
Hein, L., Obst, C., Edens, B., Remme, R.P. (2015). Progress and challenges in the development of ecosystem accounting as a tool to analyse ecosystem capital. Current Opinion in Environmental Sustainability 2014, 14; 86-92.
Herath, I., Green, S., Singh, R., Horne, D., van der Zijpp, S., Clothier, B. (2013). Water footprinting of agricultural products: a hydrological assessment for the water footprint of New Zealand's wines. Journal of Cleaner Production 41, pp. 232-243.
Hesse-Biber, S.N., Leavy, P. (2011). The Practice of Qualitative Research (2nd edition). Thousand Oaks, CA: Sage.
Hess, T.M., Lennard, A.T., Daccache, A. (2015). Comparing local and global water scarcity information in determining the scarcity footprint of potato cultivation in Great Britain. Journal of Cleaner Production 87, pp. 666-674.
Hester, E.T., Little, J.C. (2013). Measuring Environmental Sustainability of Water in Watersheds. Environmental Science and Technology 2013, 47, 8083-8090.
Hoekstra, A.Y., Chapagain, A.K., Aldaya, M.M., Mekonnen, M.M. (2011). The water footprint assessment manual: setting the global standard. Earthscan, London.
Hoekstra, A.Y. (2003). 'Virtual Water Trade'. Proceedings of the International Expert meeting on Virtual Water Trade. Value of Water Research Series No 12, UNESCO-IHE, Delft, Netherlands.
Hoekstra, A.Y., Mekonnen, M.M., Chapagain, A.K., Mathews, R.E., Richter, B.D. (2012).Global monthly water scarcity: blue water footprints versus blue water avail-ability. PLOS ONE 7, e32688.
Hoekstra, A. Y. (2016). A critique on the water-scarcity weighted water footprint in LCA. Ecological Indicators 66, pp.564-573.
Hough, M.N., Jones, R.J.A. (1997). The United Kingdom Meteorological Office rainfall and evaporation calculation system: MORECS version 2.0-an overview. Hydrology and Earth System Sciences 1(2):227-239.
Hörtenhaber, S., Piringer, G., Zollitsch, W., Lindenthal, T., Winiwarter, W. (2014). Land use and land use change in agricultural life cycle assessments and carbon footprints-the case of regionally specific land use change versus other methods. Journal of Cleaner Production 73, pp. 31-39.
Hospido, A., Núñez, M., Antón, A. (2013). Irrigation mix: how to include water sources when assessing freshwater consumption impacts associated with crops. International Journal of Life Cycle Assessment 18, pp. 881-890.
Humphries, M.M., McCann, K.S. (2013). Metabolic ecology. Journal of Animal Ecology 83, pp. 7-19.
Igos, E., Benetto, E., Baudin, I., Tiruta-Barna, L., Mery, Y., Arbault, D. (2013). Cost-performance indicator for comparative environmental assessment of water treatment plants. Science of the Total Environment 443, pp/ 367-374.
IIMM (2011). International Infrastructure Maintenance Manual International Edition 2011.
Illaszewicz, G. and Bradshaw, R. (2013). Asset management capacity development using a structural development. Water Asset Management International, Issue 3, Vol.9, 7-10.
Jaramillo, F. and Destouni, G. (2015). Local flow regulation and irrigation raise global human water consumption and footprint. Science 350 (6265), pp. 1248-1250.
Jarvis, Todd (2013). Water Scarcity: Moving beyond indexes to innovative institutions. Groundwater 51(5), pp.663-669.

Jefferies, D., Muñoz, I., Hodges, J., King, V. J., Aldaya, M., Ercin, A. E., Milà I Canals, L., Hoekstra, A. Y. (2012). Water Footprint and Life Cycle Assessment as approaches to assess impacts of products on water consumption. Key learning points from pilot studies on tea and margarine. Journal of Cleaner Production 33:155-166.

Jeswani, H.K., Azapagic, A. (2011). Water footprint: methodologies and a case study for assessing the impacts of water use. Journal of Cleaner Production 19:1288-1299.

Jones, C., Jay, S., Slinn, J., Wood, C. (2007). Environmental assessment: dominant or dormant? In: Holder, J. and McGillivray, D. (Eds.). Taking stock of Environmental Assessment: Law, Policy and Practice. Routledge-Cavendish, New York.

Kahil, M.T., Ward, F.A., Albiac, J., Eggleston, J., Sanz, D. (2016). Hydro-economic modelling with aquifer-river interactions to guide sustainable basin management. Journal of Hydrology 539, 510-514.

Karimi, P., Molden, D., Bastiaanssen, W., Xueliang, C. (2012). Water accounting to assess use and productivity of water-evolution of a concept and new frontiers. In: Geodfrey, J., Chalmers, K. (Eds). International Water Accounting: Effective Management of a Scarce Resource. Edward Elgar Publishing Inc., New York, pp. 76-90.

Karimi, P., Bastiaanssen, W., Molden, D., Cheema, M.J.M. (2013). Water Accounting Plus (WA+)- a water accounting procedure for complex river basins based on satellite measurements. Hydrologiy and Earth Systems Science 17, pp. 2459-2472.

Korhonen, Jouni (2004). Industrial ecology in the strategic sustainable development model: strategic applications of industrial ecology. Journal of Cleaner Production 12 (2004), 809-823.

Kennedy, M., Bourne-Kennedy, J. (2013). The Concise Oxford Dictionary of Music (5th edition). Oxford University Press. eISBN: 9780191727184.

Klein, J.T. (2004). Prospects for Transdsciplinarity. Futures 36, 515-526.

Klinglmair, M., Sala, S., Brandão, M. (2014). Assessing resources depletion in LCA: a review of methods and methodological issues. International Journal of Life Cycle Assessment (2014) 19:580-592. DOI 10.1007/s11367-0163-0650-9.

Klöpffer, W., Renner, I. (2008). Life cycle based sustainability assessment of products. In: Schaltegger, S., Bennett, M., Burritt, R.L., Jasch, C. (Eds). Environmental management accounting for cleaner production. Hounten (NL): Springer. P 91-102.

Kounina, A., Margni, M., Bayart, J.B., Boulay, A.M., Berger, M., Bulle, C., Frischknecht, R., Koehler, A.,Milà i Canals,L., Motoshita, M., Núñez, M., Peters, G., Pfister, S., Ridoutt, B., van Zelm, R., Verones, F., Humbert, S. (2013). Review of methods addressing freshwater use in life cycle inventory and impact assessment. International Journal of Life Cycle Assessment (2013) 18:707-721.

Krimsky, S. (2000). Transdsciplinarity for Problems at the Interstices of Disciplines. *In* Sonerville, M.A. & Rapport, D.J. (Eds). *Transidisciplinarity: ReCreating Integrated Knowledge*, 109-114. Oxford, UK: EOLSS Publishers Co.Ltd.

Kuhlman, T., Farrington, J. (2010). What is sustainability? Sustainability 2, pp. 3436-3448.

Kulig, A., Kolfoort, H., Hoekstra, R. (2010). The case for the hybrid capital approach for the measurement of the welfare and sustainability. Ecological Economics 10, pp. 118-128.

Kumara, B.K.H., Srikantaswamy, S. (2011). Environmental flow requirements in Tungabhadra River, Karnataka, India. *Nat. Resour. Res.* 20 (3), 193–205.

Kwok, C., Young, R.J., Kirkham, R.J. (2010a). Asset management of wastewater treatment projects: a review of theoretical landscape of LCC and risk based approaches. Proceedings of the Construction, Building and Real Estate Research Conference of the Royal Institution of Chartered Surveyors, Dauphine Université, Paris, 2-3 September 2010.

Kwok, C., Young, R.J., Kirkham, R.J. (2010b). Risk and asset management in privatisation/private finance of public utilities: a risk-investigation of water infrastructure projects in China. *In:* Egbu, C. (Ed). Proceedings of the 26th Annual ARCOM Conference, 6-8 September 2010. Leeds. UK. Associaition of Researcher in Construction Management, pp. 1279-1288.

Lamastra, L., Siciu, N.A., Novelli, E., Trevisan, M. (2014). A new approach to assessing the water footprint of wine: An Italian case study. Science of the Total Environment 490, pp. 748-756.

Langston, W.J., Chesman, B.S., Burt, G.R., Hawkins, S.J., Readman, J., Worsfold, P. (2003). Site characterisation of the South West European marine sites: Poole Harbour SPA. Report by Plymouth Marine Science Partnership for Environment Agency and English Nature.

Laniak, G. F.; Olchin, G.; Goodall, J.; Voinov, A.; Hill, M.; Glynn, P.; Whelan, G.; Geller, G.; Quinn, N.; Blind, M.; Peckham, S.; Reaney, S.; Gaber, N.; Kennedy, R.; Hughes, A. (2013). Integrated environmental modeling: A vision and roadmap for the future. Environmental Modelling & Software 39, pp. 3-39.

Lawrence, R.J., Després, C. (2004). Futures of Transdisciplinarity. Futures, 36, 397-405.

Leavy, Patricia (2011). Essential for Transdisciplinary Research: using problem-centred methods. Left Coast Press, INC.

Lemos, D., Dias, A.C., Gabarrell, X., Arroja, L. (2013). Environmental Assessment of an urban water system. Journal of Cleaner Production 54 (1), 157-165.

Leontief, Wassily (1951). Input Output Economics. Scientific American 185(4), 15-21.

Lifset, Reid (2013). Raising the Bar for Symbiosis, Life Cycle Assessment, and Material Flow Analysis Case Studies. Journal of Industrial Ecology (17):1, DOI: 10.1111/jiec.12003.

Lindeijer, E., Mueller-Wenl, R., Steen, B. (2002). Impact assessment of resources and land use. In: Udo de Haes HA at al (eds) Life Cycle impact assessment striving towards best practice. SETAC, Pensacola.

Little, J.C., Hester, E.T., Carey, C.C. (2016). Assessing and Enhancing Environmental Sustainability: A Conceptual Review. Environmental Science and Technology 2016, 50, 6830-6845.

Liu, J., Liu, Q., Yang, H. (2016). Assessing water scarcity by simultaneously considering environmental flow requirements, water quantity and water quality. Ecological Indicators 60, pp.434-441.

Ma, X., Xue, X., González-Mejía, A., Garland, J., Cashdollar, J. (2015). Sustainable Water Systems for the City of Tomorrow- A Conceptual Framework. Sustainability 2015, 7, 12071-12105; 10.3390/su70912071.

Macleod, C.J.A., Scholefield, D., Haygarth, P.M. (2007). Integration for sustainable catchment management. Science of the Total Environment 373, 591-602.

Maes, J., Paracchini, M.L., Zulian, G. (2011). A European assessment of the provision of ecosystem services. JRC Scientific and Technical Reports. EUR 24750. Luxembourg, Publications Office of the European Union.

Malinga, R., Gordon, L.J., Jewitt, G., Lindborg, R. (2015). Mapping ecosystem services across scales and continents- A review. Ecosystem Services 13, pp. 57-63.

Martin-Ortega, J., Balana, B.B. (2012). Cost effectiveness analysis in the implementation of the Water Framework Directive: A comparison analysis of the United Kingdom and Spain. European Water 37, pp. 15-25.

McMahon, T.A., Peel, M.C., Lowe, L., Srikanthan, R., McVicar, T.R. (2013). Estimating actual, potential, reference crop and pan evaporation using standard methodological data: a pragmatic analysis. Hydrology and Earth System Sciences 17, pp. 1331-1363.

McManus, M.C., Taylor, C.M. (2015). The changing nature of life cycle assessment. Biomass and Bioenergy 82, pp. 13-26.

Manzardo, A., Loss, A., Fialkiewicz, W., Rauch, W., Scipioni, A. (2016). Methodological proposal to assess the water footprint accounting of direct water use at an urban level: a case study of the Municipality of Vicenza. Ecological Indicators 69:165-175.

Maxwell, D., McKenzie, E., Traldi, R., 2014. Natural Capital Coalition; valuing natural capital in business. ICAEW.

Mekonnen, M.M. and Hoekstra, A.Y. (2015). Global grey water footprint and water pollution levels related to anthropogenic nitrogen loads to fresh water. Environmental Science and Technology 49, pp. 22860-12868. DOI: 10.1021/acs.est.5b03191.

Men, B., Yu, T., Kong, F., Yin, H. (2014). Study on the minimum and appropriate in stream ecological flow in Yitong River based on Tennant method. *Nat. Environ.Pollut. Technol.* 13 (3), 541–546.

Milà i Canals, L., Chapagain, A., Orr, S., Chenoweth, J., Anntont, A., Clift, R., 2010. Assessing freshwater use impacts in LCA, part 2: case study of broccoli production in the UK and Spain. International Journal of Life Cycle Assessment (2010) 15:598-607.

Miller, T.R., Wiek, A., Sarewitz, D., Robinson, J., Olsson, L., Kriebel, D., Loorbach, D. (2014). The future of sustainability science: a solutions-oriented research agenda. Sustainability Science 9, pp. 239-246.

Molden, D., Skathivadivel, R. (1999). Water accounting to assess use and productivity of water. International Journal of Water Resources Development 15 (1-2), pp. 55-71.

Momblanch, A., Andreu, J., Paredes-Arquiola, J., Solera, A., Pedro-Monzonís, M. (2014). Adapting water accounting for integrated water resource management. The Júcar Water Resource System (Spain). Journal of Hydrology 519, pp. 3369-3385.

Morais, S.A., Delerue-Matos, C. (2010). A perspective on LCA application in site remediation services: critical review of challenges. Journal of Hazardous Materials 175, pp. 12-22.

Morales-Pinzón, T., Rieradevall, J., Gasol, C.M., Gabarrell (2012). Potential of rainwater resources based on ubran and social aspects in Colombia. Water and Environmental Journal, 26 (4), 550-559.

Morera, S., Corominas, Ll., Poch, M., Aldaya, M.M., Comas, J. (2016). Water footprint assessment in wastewater treatment plants. Journal of Cleaner Production 112:4741-4748.

Morgan, R.K. (2012). Environmental Impact Assessment: the state of the art. Impact Assessment and Project Appraisal 30 (1), pp. 5-14.

Mouri, G., Takizawa, S., Fukushi, K., Oki, T. (2013a). Estimation of the effects of chemically treatment urban sewage system based on life cycle management. Sustainable Cities and Society 9, pp. 23-31.

Mouri, G., Shinoda, S., Taikan, O. (2013b). Assessment of the historical environmental changes from a survey of local residents in an urban-rural catchment. Ecological Complexity 15, pp. 83-96.

Müller-Wenk, R., Brandão, M., 2010. Climatic impact of land use in LCA-carbon transfers between vegetation/soil and air. International Journal of Life Cycle Assessment (2010) 15:172-182.

Nafi, A., Bentarzi, Y., Granger, D., Cherqui, F. (2014). Eco-EAR: A method for the economic analysis of urban water systems providing services. Urban Water Journal, 1-15.

Narangalla, R., Trotter, N. 2006. LCA in Asset Management-A metropolitan water company's experiences. Report. Accessed 3rd of December 2013.

National River Flow Archive (NRFA). Webpage: http://nrfa.ceh.ac.uk/. Accessed on the 27th of May 2016.

Natural Capital Initiative (2015). Valuing our Life Support Systems 2014. Summit Summary Report.

Nebbia, G. (1975). La matrice de rifiuti. Rassegna Economics, 39(1), 37-62.

Nebbia, G. (2000). Contabilità monetaria e contabilità ambientale. Economic Pubblica, 30(6), 5-33.

Niazi, A., Prasher, S.O., Adamowski, J., Gleeson, T. (2015). A System Dynamics model to conserve arid region water resources through aquifer storage and recovery in conjunction with a dam. Water (6):2300-2321.

Niero, M., Pizzol, M., Bruun, H.G., Thomsen, M. (2014). Comparative life cycle assessment of wastewater treatment in Denmark including sensitivity and uncertainty analysis. Journal of Cleaner Production 68, pp. 25-35.

Nix, John (2015). Farm Management Pocketbook. 45th edition. ISBN 978-0-9576939-1-3.

NRC (2014). Sustainability Concepts in Decision-Making: Tools and Approaches for the US Environmental Protection Agency; National Research Council: Washington, DC, p 156.

Norton, M., Lane, A. (2012). 'New Water Architecture': an integrated water management model. Management, procurement and Law, vol. 165, Issue MP3, pp. 159-171.

Núñez, M., Oliver-Solà, J., Rieradevall, J., Gabarrell, X. (2010). Water Management in Integrated Service Systems: Accounting for Water Flows in Urban Areas. Water Resources Management 24:1583-1604.

Núñez, M., Bouchard, C.R., Bulle, C., Boulay, A. and Amrgni, M. (2016). Critical analysis of life cycle impact assessment methods addressing consequences of freshwater use on ecosystems and recommendations for future method development. International Journal of Life Cycle Assessment 21: 1799-1815. doi:10.1007/s11367-016-1127-4.

Obst, C., Vardon, M. (2014). Recording environmental assets in national accounts. Oxford Review of Economic Policy 30 (1), 126-144.

Ody, K., Martineau, H. (2015). Catchment Management Strategy: Scoping study report for nitrogen reduction. Report for Wessex Water from Ricardo-AEA. ED 60948, Issue Number 1.

OFWAT (2006). The Development of the Water Industry in England and Wales. OFWAT, Birmingham.

OFWAT (2015a).Reliable services for customers- consultation on Ofwat's role on resilience. July 2015.

OFWAT (2015b). Trust in Water, Water 2020: Regulatory framework for wholesale markets and the 2019 price review. December 2015. ISBN 978-1-910739-28-0.

Oliver-Solà, J., Armero, M., Martinez de Foix, B., Rieadevall, J. (2013). Energy and environmental evaluation of municipal facilities: Case study in the province of Barcelona. Energy Policy 61, 920-930.

Ownes, J.W. (2002). Water resources in life cycle impact assessment- considerations in choosing category indicators. Journal of Industrial Ecology 5(2):37-54.

O' Riordan, T., Voisey, H. (1997). The Political Economy of Sustainable Development Environmental Politics 1.

Palmer, A.M., Kiu, J., Matthews, J.H., Mumba, M., D'Odorico, P. (2015). Water security: Gray or green? Science 349 (6248), pp., 584-585.

Palmer, S.J. (2010). Future challenges to asset investment in the UK water industry: the wastewater asset investment risk mitigation offered by minimising principal operating cost risks. Journal of Water and Climate Change 1(1), pp. 17-35.

Papajorgji, Petraq J and P. M Pardalos. *Software Engineering Techniques Applied To Agricultural Systems*. 2nd ed. Springer, 2014. ISBN 978-1-4899-7463-1.

Pastor, A.V., Ludwig, F., Biemans, H., Hoff, H., Kabat, P. (2014). Accounting for environmental flow requirements in global water assessments. *Hydrol. Earth Syst.Sci*. 18 (12), 5041–5059.

Paterson, W., Rushforth, R., Rudell, B.L., Konar, M., Ahams, I.C., Gironás, J., Mijic, A., Mejia, A. (2015). Water Footprint of Cities: A Review and Suggestions for Future Research. *Sustainability* 7, 8461-8490; 10.3390/su7078461.

Pedro-Monzonís, M., Jiménez-Fernández, P., Solera, A., Jiménez-Gavilán, P. (2016a). The use of AQUATOOL DSS applied to the System of Environmental-Economic Accounting for Water (SEEAW). Journal of Hydrology 533 (2016) 1-14.

Pedro-Monzonís, M., Solera, A., Ferrer, J., Estrela, T, Paredes-Aquiola (2016b). Water accounting for stressed river basins based on water resources management models. Science of the Total Environment 565:181-190.

Pérez-Maqueo, P., Martinez, M.L., Vázquez, G., Equihua, M. (2013). Using Four Capital to Assess Watershed Sustainability. Journal of Environmental Management (2013) 51:679-693; 1.1007/s00267-012-9972-9.

Perrin, A., Basset-Mens, C., gabrielle, B. (2014). Life cycle assessment of vegetable products: a review focusing on cropping systems diversity and the estimation of field emissions. International Journal of Life Cycle Assessment 19:1247-1263.

Perry, C. (2014). Water footprints: Path to enlightenment or false trail? Agricultural Water Management 134, pp. 119-125.

Pfister, S., Koehler, A., Hellweg, S. (2009). Assessing the Environmental Impacts of Freshwater Consumption in LCA. Environmental Science and Technology 2009, 43, 4098-4104.

Pfister S, Saner D, Koehler A (2011) The environmental relevance of water consumption in global power production. International Journal of Life Cycle Assessment 16, pp. 580–591

Pfister S., Bayer P. (2014) Monthly water stress: spatially and temporally explicit consumptive water footprint of global crop production. Journal of Cleaner Production 73, pp. 52–62.

Pfister, S.,Vionnet, S., Levova, T., Humbert, S. (2015). Ecoinvent 3: assessing water use in LCA and facilitating water footprinting. International Journal of Life Cycle Assessment 21, pp. 1349-1360.

Poff, N. L., Richter, B. D., Arthington, A. H., Bunn, S. E., Naiman, R. J., Kendy, E., Acreman, M., Apse, C., Bledsoe, B. P., Freeman, M. C., Henriksen, J., Jacobson, R. B., Kennen, J. G., Merritt, D. M., Oâkeeffe, J. H., Olden, J. D., Rogers, K., Tharme, R. E., Warner, A. (2010). The ecological limits of hydrological alteration (ELOHA): a new framework for developing regional environmental flow standards. *Freshwater Biology* 55(1), 147-170.

Pohl, C. (2008). From science to policy through transdisciplinary research. Environmental Science and Policy 11 (2008), 46-53. ; 10.1016/j.envsci.2007.06.001.

Prato, T., Herath, G. (2007). Multi-criteria decision analysis for integrated catchment management. Ecological Economics 63, pp/ 627-632.

Ran, Y., Lannerstad, M., Herrero, M., Van Middelaar, C.E., De Boer, I.J.M. (2016). Assessing water resource use in livestock production: A review of methods. Livestock Science 187, pp. 68-79.

Rajagopal, D. (2016). A step towards a general framework for consequential life cycle assessment. Methods, Tools and Software, DOI: 10.1111/jiec.12433.

Rebitzer, G., Hunkeler, D. (2003). Life cycle costing in LCM: ambitions, Opportunities and Limitations. Discussing a framework. International Journal of Life Cycle Assessment 8 (5), pp. 253-256.

Richter, B.D., Mathews, R., Wigington, R. (2003). Ecologically sustainable water management: managing river flows for ecological integrity. *Ecological Applications* 13, 206-224.

Richter, B.D. (2010). Re-thinking environmental flows: From allocations and reserves to sustainability boundaries. *River Research and Applications* 26(8):1052-1063.

Ridoutt, B.G., Pfister, S. (2013). A new water footprint calculation methods integrating consumptive and degradative water use into a single stand-alone weighted indicator. International Journal of Life Cycle Assessment 18, pp. 204-207.

Ridoutt, B.G., Page, G., Opie, K., Huang, J., Bellotti, W. (2014). Carbon, water and land use footprints of beef and cattle production systems in southern Australia. Journal of Cleaner Production 73 (2014):24-30.

Risch, E., Loubet, P., Núñez, M., Roux, P. (2014). How environmentally significant is water consumption during wastewater treatment?: Application of recent developments in LCA to WTT technologies used at 3 contrasted geographical locations. Water Research 57, pp. 20-30.

Rouse, Michael (2013). Institutional Governance and Regulation of Water Services: The essential elements (2nd edition). IWA Publishing.

Rudell, B.L., Adams, E.A., Rushforth, R., Tidwell, V.C. (2014). Embedded resource accounting for coupled natural-human systems: An application to water resource impacts of the western U.S. electrical energy trade. *Water Resources Research* 50, 7957-7972.; 10.1002/2013WR014531.

Ruviano, C.F., de Leis, C.M., Lampert, Vd.N., Barcellos, J.O.J., Dewes, H. (2014). Carbon footprint in different beef production systems on a southern Brazilian farm: a case study. Journal of Cleaner Production 96, pp. 435-443.

Sadhukhan, J. (2014). Distributed and micro-generation from buigas and agricultural application of sewage sludge: Comparative environmental performance analysis using life cycle approaches. Applied Energy 122, pp. 196-206.

Sala, S., Farioli, F., Zamagni, A. (2013a). Life cycle sustainability assessment in the context of sustainability science progress. International Journal of Life Cycle Assessment 18, pp. 1686-1697.

Sala, S., Farioli, F., Zamagni, A. (2013b). Progress in sustainability science: lessons learnt from current methodologies for sustainability assessment. International Journal of Life Cycle Assessment 18, pp. 1653-1672.

Sanga, G.J., Mungatana, E.D. (2016). Integrating ecology and economics in understanding responses in securing land-use externalities internalization in water catchment. *Ecological Economics* 121:28-39.

Schwaninger, M., Ulli-Beer, S., Kaufmann-Hayoz, R. (2007). Policy Analysis and Design in Local Public Management: A system dynamics approach. In: Handbook of Transdisciplinary Research. Hadorn, G.H. et al (Eds). Springer, p. 205-221.

Settanni, E., Newnes, L.B., Wright, J. (2015). Fifty shades of greywater: an outcome perspective on wastewater systems operations' performance. Conference Proceedings of the 19th Cambridge International Manufacturing Symposium, 24-25 September 2015. ISBN 978-1-902546-67-4.

Settanni, E., Newnes, L.B., Thenent, N.E., Parry, G., Goh, Y.M., (2014). A through-life costing methodology for use in product-service-systems. *International Journal of Production Economics* 153, pp. 161-177; 10.1016/j.ijpe.2014.02.016.

Settanni, E., Notarnicola, B., Tassielli, G. (2012a). Life Cycle Assessment (LCA). The Berkshire Encyclopedia of Sustainability. Berkshire Publishing, pp. 221-224.

Settanni, E., Notarnicola, B., Tassielli, G. (2012b). Life Cycle Costing (LCC). The Berkshire Encyclopedia of Sustainability. Berkshire Publishing, pp. 225-227.

Šerifi, V., Dašić, P., Ječmenica, R., Labović, D. (2009). Functional and Information Modelling of Production Using IDEF Methods. *Journal of Mechanical Engineering* 55 (2), 131-140.

Slagstad, H., Brattebø, H. (2014). Life cycle assessment of the water and wastewater system in Trondheim, Norway-A case study. Urban Water Journal, Vol.11, No.4, 323-334. doi:10.1080/1573062X.2013.795232.

Smakhtin, V.U., Shilpakar, R.L., Hughes, D.A. (2006). Hydrology-based assessment of environmental flows: an example from Nepal. *Hydrol. Sci. J.* 51 (2), 207–222.

Smith, R.N.B., Blyth, E.M., Finch, J.W., Goodchild, S., Hall, R.L., Mary, S. (2006). Soil state and surface hydrology diagnosis based on MOSES in the Met Office Nimrod nowcasting system. Meteorological Applications 13; pp. 89-109.

Soussana, Jean-François (2014). Research priorities for sustainable agri-food systems and life cycle assessment. Journal of Cleaner Production 73, pp. 19-23.

Stasinopoulos, P.; Smith, M. H.; Hargroves, K. C.; Desha, C. (2009). Whole System Design: An Integrated Approach to Sustainable Engineering. Journal of Education for Sustainable Development 3 (2), pp. 241–243.

Stavridou, I., Ferreira, A. (2010). Multi- Inter- and Trans- disciplinary research promoted by the European Cooperation in Science and Technology (COST): Lessons and experiments. Research Report. inria-00512712v2.

Stoeglehner, G., Edwards, P., Daniels, P., Narodoslawsky, M. (2011). The water supply footprint (WSF): a strategic planning tool for sustainable regional and local water supplies. Journal of Cleaner Production 19:1677-1686.

Stodden, V., McNutt, M., Bailey, D.H., Deelman, E., Gil, Y., Hanson, B., Heroux, M.A., Ioannidis, JPA, Taufer, M. (2016). Enhancing reproducibility for computational models. Science 354 (6317), pp. 1240-1241.

Suh, Sangwon and Kagawa, Shigemi (2005). Industrial Ecology and Input-Output Economics: an Introduction. Economic Systems Research, Vol. 17, No .4, 349-364.

Swarr, T.E., Hunkeler, D., Klöpffer, W., Pesonen, H.L., Ciroth, A., brent, A.C., Pagan, R. (2011). Environmental Life Cycle Costing: A Code of Practice. Society of Environmental Toxicology and Chemistry (SETAC). p.92.

Tennant, D.L. (1976). Instream flow regimens for fish, wildlife, recreation and related environmental resources. Fisheries 1, 6–10.

Termes-Rifé, M., Molinos-Senante, M., Harnández-Sancho, F., Sala-Garrido, R. (2013). Life Cycle Costing: a tool to manage the urban water cycle. Journal of Water Supply: Research and Technology-AQUA 62.7 2013, 468-476. doi:10.2166/aqua2013.110.

Thaler, S., Zessner, M., Bertran De Lis, F., Kreuzinger, N., Fehringer, R. (2012). Considerations on methodological challenges for water footprint calculations, Water Science and Technology 65, pp/ 1258-1264.

Tharme, R.E. (2003). A global perspective on environmental flow assessment: emerging trends in the development and application of environmental flow methodologies for rivers. River Res. Appl. 19, 397–441.

Thomspon, N., Barrie, I.A., Ayles, M. (1981). The Meteorological Office rainfall and evaporation calculation systems: (MORECS). Hydrological Memories 45, Meteorological Office, Bracknell.

Tillotson, M.R., Liu, J., Guan, D., Wu, P., Zhao, X., Zhang, G., Pfister, S., Pahlow, M. (2014). Water Footprint Symposium: where next for water footprint and water assessment methodology? International Journal of Life Cycle Assessment (2014) 19:1561-1565.

TIlmant, A., Marques, G., Mohamen, Y. (2015). A dynamic water accounting framework based on marginal resource opportunity cost. Hydrology and Earth Systems Science Discussion 19, pp. 1457-1467.

Toit, D.D. (2005). Preparing people for Integrated Catchment Management: a proposed learning alliance for the implementation of a new legal framework for water management in South Africa; “reflective learning in context”. Proceedings of the Symposium on learning alliance for scaling up innovative approach in the water and sanitation sector. 7-9 June 2005, Delft, the Netherlands, p229-242.

Too, E.G. (2011). Strategic Infrastructure Asset Management: The way forward. *In:* J.Mathew et al. (eds.): Engineering Asset Management and Infrastructure Sustainability. Springer-Verlag, London.

Uche, J., Martínez-Gracia, A. and Carmona, U. (2013). Life Cycle Assessment of the supply and use of water in the Segura Basin. International Journal of Life Cycle Assessment 19.3; 688-704.

UKWIR. (2014). Environmental impact assessment to compare the benefits of achieving tight BOD standards versus increase in whole lifecycle carbon. Report Ref. No. 14/WW/20/7.

UNEP/SETAC (2011). Towards a life cycle sustainability assessment. UNEP/SETAC, Paris, p.65.

UNEP (2011). Water Footprint and Corporate Water Accounting for Resource Efficiency.

United Nations Educational, Scientific and Cultural Organisation (UNESCO) (2014). World Water Development Report: Water and Energy.

United Nations Environmental Programme, UNEP (2012). The Natural Capita Declaration. UNEP Financial Initiative: Innovative financing for sustainability. Global Canopy Programme.

United Nations Statistic Division, UNDS (2012). System of Environmental-Economic Accounting for Water (SEEAW). United Nations, Department of Economic and Social Affairs, Statistics Division, (United Nations publication, Sales No. E.11.XVII.12), New York, 2012.

United Nations World Water Assessment Programme (2012). The United Nations World Water Development Report 4: Managing Water under Uncertainty and Risk. Paris, UNESCO.

United Nations Convention on Wetlands of International Importance especially as Waterfowl Habitat. Ramsar (Iran), 2 February 1971. UN Treaty Series No. 14583.

United Nations Economic Commission for Europe (2009). Measuring Sustainable Development.

Valdivia, S., Ugaya, C.M.L., Hildenbrand, J., Traverso, M., Mazijn, B., Sonnemann, G. (2013). A UNEP/SETAC approach towards a life cycle sustainability assessment- our contribution to RIO+20. International Journal of Life Cycle Assessment 18, pp. 1673-1685.

Valipour, M., Sefidkouhi, M.A.G., Eslamian, S. (2015). Surface irrigation simulation models: a review. International Journal of Hydrology, Science and Technology, 5 (1), 51-70.

Vanham, D., Bidoglio, G. (2013). A review of the indicator water footprint for the EU28. Ecological Indicators 26, pp. 61-75.

van Heck, G.J. (2010). Waternet drinking water tactical planning process. Proceedings of the Institution of Civil Engineers, Issue ME4, pp. 247-256. doi:10.1680/muen.2010.163.4.247.

van Hoof, G., Buyle, B., Kounina, A., Humbert, S. (2013). Life Cycle-based Water Assessment of a Hand Dishwashing Product: Opportunities and Limitations. Integrated Environmental Assessment and Management 9(4), pp. 633-644.

van der Werf, H.M.G., Garnett, T., Corson, M.S., Hayashi, K., Huisingh, D., Cederberg, C. (2014). Towards eco-efficient agriculture and food systems: theory, praxis and future challenges. Journal of Cleaner Production 73, pp. 1-9.

Van der Werf, H.M.G., Corson, M.S., Wilfart, A. (2013). LCA Food 2012-towards sustainable food systems. International Journal of Life Cycle Assessment 18, pp. 1180-1183.

van der Velder, J., Klatter, L., Bakker, J., 2013. A holistic approach to asset management in the Netherlands. Structure and Infrastructure Engineering 9(4), pp. 340-348.

van der Voet, Esten (2011). Metabolic Effects of Transitions: An Industrial-Ecology Based Exploration. Journal of Industrial Ecology, Special Issue: Visions for Industrial Ecology, pp. 646-648.

Venkatesh, G. and Brattebø, H. (2010) Environmental analysis of chemicals and energy consumption in water treatment plants: case study of Oslo, Norway. Water Science and Technology – Water Supply 12(2), pp. 200–211.

Vicente, D.J., Rodríguez-Sinobas, L., Garrote, L., Sánchez, R. (2016). Application of the system of environmental economic accounting for water SEEAW to the Spanish part of the Duero basin: Lessons learned. Science of the Total Environment 563-564, pp.611-622.

Vince, F. (2007). Proposition for the LCI framework. Unpublished. *In:* LCA Compendium – The Complete World of Life Cycle Assessment. Special Types of Life Cycle Assessment.

Vionnet, S., Lessard, L., Offutt, A., Levova, T., Humbert, S. (2012). Quanits water database- technical report Quantis International, Lausanne.

Vogtländer, J.G., Beatens, B., Bijma, A., Brandjes, E., Lindeijer, E., Segers, M., Witte, F., Brezet, J.C., Hendriks, Ch.F. (2010). LCA-based Assessment of Sustainability: the Eco-costs/Value Ratio (EVR). VSSD, Delft, The Netherlands.

Vörösmarty, C.J., Hoekstra, A.Y., Bunn, S.E., Conway, D., Gupta, J. (2015). Water scale for water governance? Science 349 (6247), p.478.

Walter, A.I., Wiek, A., Scholz, R.W. (2007). Constructing Regional Development Strategies: A case study approach for integrated planning and synthesis. *In*: Handbook of Transdisciplinary Research. Hadorn, G.H. et al (Eds). Springer, p 223-243.

Wang, Z., Huang, K., Yang, S., Yu, Y. (2013). An input-output approach to evaluate the water footprint and virtual water trade of Beijing, China. Journal of Cleaner Production 42, pp. 172-179.

Ward, F.A., Pulido-Velazquez, M. (2009). Incentive pricing and cost recovery at the basin scale. Journal of Environmental Management 90, pp. 293-313.

Water Act (1973). Parliament of the United Kingdom. 1973-07-18.

Water Act (1989). Parliament of the United Kingdom. 1989-07-06.

Water Act (2014). Parliament of the United Kingdom. 2014-05-14.

WCED (1987). World Commission on Environment and Development, Our Common Future. Oxford University Press, New York.

Wessex Water (2014). Water-the way ahead 2015-2040. Wessex Water's long-term strategy.

Wichelns, Dennis (2015). Virtual water and water footprints do not provide helpful insight regarding international trade or water scarcity. Ecological Indicators 52, 277-283.

Weidema, B.P. (2009). Using the budget constraint method to monetise impact assessment results. Ecological Economics 68 (6), pp. 1591-1598.

Wickson, F., Carew, A.L., Russell, A.W. (2006). Transdisciplinary research: characteristics, quandaries and quality. Futures, 38 (9), 1046-1059

Wiedmann, T., Wilting, H., Lenzen, M., Lutter, S., Palm, V. (2011). Quo Vardis MRIO? Methodological, data and institutional requirements for multi-region input-output analysis. Ecological Economics 70 (2011) 1937-1945.

Williams, B. and Hummelbrunner, R. (2011). *Systems Concepts in Action*. 1st ed. Stanford, Calif.: Stanford Business Books.

Wilson, E.M. (1990). Engineering Hydrology, Fourth Edition. MacMillan, London.

Wiesner, C.J. (1970). Climate irrigation and agriculture. Angus and Robertson, Sydney.

Wood, C.M. (2003). Environmental Impact Assessment: A comparative review. 2nd edition. Harlow Prentice Hall.

Woodhouse, J. (2006). Putting the jigsaw puzzle together: PAS 55 standard for the integrated, optimised management of assets. 21st International Maintenance Conference, 5-8 September 2006, Daytona Beach Florida. IMC.

Xu, X., Schoen, M.E., Ma, X., Hawkins, T.R., Ashbolt, N.J., Cashdollar, J., Garland, J. (2015). Critical insights for a sustainability framework to address integrate community water services: Technical metrics and approaches. Water Research 77 (2015), 155-169.

Xue, X., Pang, Y., Landis, E. (2014). Evaluating agricultural management practices to improve the environmental footprint of corn-derived ethanol. Renewable Energy 66, pp. 454-460.

Yang, Y. (2016). Two sides of the same coin: consequential life cycle assessment based on the attributional framework. Journal of Cleaner Production 127, pp. 274-281.

Yang, H., Pfister, S., Bhaduri, A. (2013). Accounting for a scarce resource: virtual water and water footprint in the global water system. Current opinion in Environmental Sustainability 2013, 5:599-606.

Yin, R.K. (1993). Applications of case study research. Applied Social Research Methods Series, Volume 34. Sage Publications.

Yin, R.K. (2009). Case Study Research: Design and Methods (4th edition). Thousand Oaks, CA: Sage.

Yoshido, H.., Christensen, T.H., Scheutz, C. (2014). Life cycle assessment of sewage sludge management: a review. Water Management & Research 31, pp. 1083-1101.

Yu, Y., Hubacek, K., Feng, K., Guan, D. (2010). Assessing regional and global water footprints for the UK. Ecological Economics 69, 1140-1147.

Zaccai, E. (2012). Over two decades in pursuit of sustainable development: Influence, transformations, limits. Environmental Development 1 (1), pp. 79–90.

Ziolkowska, J. R., Ziolkowski, B. (2016). Effectiveness of Water Management in Europe in the 21st Century. Water Resources Management 30 (7), 2261-2274.; 10.1007/s11269-016-1287-9.

Zeng, Z., Liu, J., Savenije, H.H.G. (2013). A simple approach to assess water scarcity integrating water quantity and quality. Ecol. Indic. 34, 441–449.

Zhang, Y., Huang, K., Yu, Y., Hu, T., Wei, J. (2015). Impact of climate change and drought regime on water footprint of crop production: the case of Lake Dianchi Basin, China. Natural Hazards 79, pp. 549-566.

Zhang, G.P., Mathews, R.E., Frapporti, G., Mekonnen, M.M. (2014). Water Footprint Assessment for the Hertfordshire and North London Area. Water Footprint Network & Environment Agency. Report-RESE0003335.

Zhang, Y., Baral, A., Bakshi, B.R. (2010). Accounting for Ecosystem Services in Life Cycle Assessment, Part II: Toward an Ecologically Based LCA. Environmental Science and Technology 44, pp. 2624-2631.

Zhao, X., Yang, H., Yang, F., Chen, B., Qin, Y. (2010). Applying the input-output method to account for water footprint and virtual water trade in the Haihe River Basin in China/ Environmental Science and Technology, 44, 9150-9156.

Zhi, Y., Yang, F., Yin, X.A. (2014). Decomposition analysis of water footprint changes in a water-limited river basin: a case study of the Haihe River basin, China. Hydrology and Earth System Science, 18, 1549-1559.

Zonderland-Thomassesn, M.A., Lieffering, M., Ledgard, S.F. (2014). Water footprint of beef cattle sheep produced in New Zealand: water scarcity and eutrophication impacts. Journal of Cleaner Production 73, pp. 253-262.

APPENDIX A

APPENDIX A presents the data used to computed the natural water budget of the Poole Harbour Catchment. Processed data from the Met Office (MORECS data) and the National River Flow Archive (NRFA) are illustrated, as indicated in the table captions below.

Table A1: MORECS data on rainfall, potential and actual evaporation and effective precipitation for squares 180 and 181, representing 52% and 48% of the catchment values respectively. Data computed for an annual time scale.

	Annual Rainfall		**Annual Potential Evap**		**Annual Actual Evap**		**Annual Effective Precip**	
Year	Annual SUMS for 180	Annual SUMS for 181	Annual SUMS for 180	Annual SUMS for 181	Annual SUMS for 180	Annual SUMS for 180	Annual SUMS for 180	Annual SUMS for 180
1996	437.6	479.7	437.1	423.8	405	411.4	84.5	123.5
1997	829.0	846.3	583.1	561.6	642.9	517.4	409.4	314.7
1998	964.1	976.6	584.7	573.9	566.2	542.4	404.3	435.3
1999	989.2	982.6	636.9	559.1	550.2	507.7	444.2	473.1
2000	1120.7	1219.1	561.9	534.6	650	500.9	593.7	718.5
2001	838.0	884.8	543.9	500.6	498.4	446.4	341.7	441.3
2002	1110.3	1141.6	564.7	548.5	551.3	530.2	566.5	610
2003	720.4	757.7	602.1	577.9	466.9	452.9	265.7	305.1
2004	815.4	826.3	571.1	560.4	538.9	509.6	281.6	322.5
2005	747.3	704.4	572.6	567.7	532.5	506.8	222.3	201.1
2006	734.1	773.9	608	610.9	505.3	492.4	238.7	274.8
2007	970.7	956.4	586.1	583.4	566.3	550.5	407	405
2008	1001.7	994.3	571.5	568	564.1	554.8	493	478
2009	934.9	875.9	559.7	582.2	527.3	517.3	412.1	356.4
2010	758.0	764.1	543.6	547.4	468.2	434.1	300.1	340.7
2011	745.3	747.7	576.9	579.1	522.5	512.2	231.7	244.3
2012	1224.7	1197.5	535	537.3	525.2	515.3	703	664.2
2013	945.6	992.2	583.3	581.6	481.8	471.5	471.4	521.3
2014	1173.8	1233.7	611.4	613.7	586.5	559.1	594.9	674.4
2015	868.3	867.9	587.6	595.8	551	538.9	324.4	328.4
Square Average	873.7	890.2	550.7	540.3	516.4	486.1	385.1	408.9
Annual Catchment Average	**881.6**		**545.7**		**501.9**		**396.5**	

Table A2: MORECS data on rainfall, potential and actual evaporation and effective precipitation for squares 180 and 181, representing 52% and 48% of the catchment values respectively. Data computed for the dry season: April to September.

Dry Season: April-September	**Seasonal Rainfall**		**Seasonal Potential Evap**		**Seasonal Actual Evap**		**Effective Precip**	
Year	Seasonal SUMS for 180	Seasonal SUMS for 181	Seasonal SUMS for 180	Seasonal SUMS for 181	Seasonal SUMS for 180	Seasonal SUMS for 181	Seasonal SUMS for 180	Seasonal SUMS for 181
1996	185.2	204.9	349.3	341.8	318.7	331.7	11.1	20.1
1997	403.3	352.6	442.7	436	504.1	394.7	31	38.6
1998	446.9	429.1	425.6	423.4	408.6	393.6	95.1	93.6
1999	478.5	451.3	424.7	416.7	395.4	367.1	87.7	100.7
2000	455.3	477.7	399.4	388.6	488.3	356.3	135.9	172.7
2001	329.5	291	414.7	379.8	370.8	327.6	56.6	61.8
2002	363.2	375.2	396.6	394.7	392.4	385.1	47.2	56.6
2003	245.6	244.9	460.6	444	343.6	337.4	13	22.9
2004	351	338.7	428.8	423	398.1	374.4	43.6	65.3
2005	329.0	296.0	424.8	423.1	386.8	365.2	51.6	40.8
2006	262.1	272.6	457.2	466	355.9	350.9	33.1	48.6
2007	450.9	427.2	432.7	437	414.7	406.9	70.3	61.5
2008	493.3	499.2	428.2	432.7	421.8	421	140.1	141.6
2009	341.4	321.3	424.8	442.5	395.1	384.2	22	30.9
2010	311.7	273.9	428.3	430.9	354.7	319.6	48.6	54.5
2011	353.4	351.7	435.9	441.1	384.8	379.3	20.6	35.3
2012	657.5	632	397	398.4	386.5	376.6	258	254
2013	240.6	244.9	438.6	438.9	338.6	331.5	14.2	32.7
2014	416.9	415.9	460.4	464.5	437.5	412.9	77.5	116
2015	430.6	433.1	429.3	436.3	393.4	382.2	46.3	63.2
Square Average	377.3	366.7	424.98	422.97	394.49	369.91	65.175	75.57
Seasonal Average	**372.2**		**424.0**		**382.7**		**70.2**	

Table A3: MORECS data on rainfall, potential and actual evaporation and effective precipitation for squares 180 and 181, representing 52% and 48% of the catchment values respectively. Data computed for the wet season: October to March.

Wet Season: October-March	**Seasonal Rainfall**		**Seasonal Potential Evap**		**Seasonal Actual Evap**		**Effective Precip**	
Years	Seasonal SUMS for 180	Seasonal SUMS for 181	Seasonal SUMS for 180	Seasonal SUMS for 181	Seasonal SUMS for 180	Seasonal SUMS for 181	Seasonal SUMS for 180	Seasonal SUMS for 181
1996-1997	404.7	436.3	145.6	133.3	143.9	130.7	171.7	210.2
1997-1998	472.7	559	158.9	143.9	157.3	140.6	405.7	328.5
1998-1999	513.1	521.9	151.4	148	149.5	146.4	316	322.6
1999-2000	472.2	503.3	214	136.4	156.6	134.3	313.4	347.3
2000-2001	787.2	910.9	138.8	127.4	138.4	126.7	602.5	733.1
2001-2002	496.6	508.2	150.7	138	152.5	139.1	241.9	262.9
2002-2003	657.8	674.6	164.3	151.4	151.5	139.5	460.7	490.8
2003-2004	524.6	567.2	145.8	142.7	127.6	124.3	282.4	318.2
2004-2005	369.9	401.7	144.6	141.1	143.1	138.6	136.3	157.3
2005-2006	434.1	406.3	141.2	137.2	139.3	133.9	194.2	166.3
2006-2007	619.2	683.1	167.5	160.7	166	157.6	348.2	410
2007-2008	480.5	481	146.9	139.1	145.2	136.4	297.2	291.2
2008-2009	486.1	474.8	130.5	125.9	129.1	123.4	355.1	342.4
2009-2010	564.8	531.8	124.5	129.8	122	123.9	356.5	293.9
2010-2011	419.3	462.6	120.3	118.1	118.5	115.8	233.2	262.8
2011-2012	328.5	320.9	149.4	150.9	146.4	146	140.6	118
2012-2013	739.5	743.5	130.8	130.5	131.1	130.8	611.5	596.6
2013-2014	842.7	918.9	157.4	156.5	156.1	154	582.5	645.2
2014-2015	549.7	550.7	157.4	150.3	155.3	146.3	311.7	303.2
2015-2016	583.8	637.5	158.6	162.4	158.1	160.4	412.5	450.3
Square Average	537.4	564.7	149.93	141.18	144.375	137.435	338.69	352.54
Seasonal Average	**550.5**		**145.7**		**141.0**		**345.3**	

Tables A4(a), (b): Environmental Flow Requirements of the two main rivers of the Poole Harbour Catchment: Frome and Piddle. Computations performed for 'good' ecological status, according to the Liu et al. (2016) method. Percentage of mean annual flow (*Pij*) assumed as *Pij*= 0.2 for the period October to March and as *Pij*=0.4 for the period April to September. Units in cubic meter (m^3) of water.

(a)	**FROME**																			
	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	
Jan	5.6E+06	2.4E+06	9.5E+06	8.6E+06	6.4E+06	8.7E+06	3.1E+06	9.3E+06	6.3E+06	3.4E+06	3.0E+06	7.5E+06	7.0E+06	4.9E+06	6.2E+06	4.9E+06	2.6E+06	9.2E+06	1.1E+07	
Feb	5.9E+06	3.6E+06	4.2E+06	4.8E+06	4.4E+06	7.0E+06	5.5E+06	5.5E+06	5.7E+06	2.6E+06	2.7E+06	6.1E+06	4.9E+06	6.6E+06	5.0E+06	3.9E+06	2.1E+06	7.0E+06	9.4E+06	
Mar	5.0E+06	4.6E+06	4.1E+06	3.9E+06	4.1E+06	7.1E+06	4.9E+06	4.5E+06	4.0E+06	2.4E+06	3.2E+06	7.5E+06	4.3E+06	5.0E+06	5.2E+06	3.3E+06	2.1E+06	5.9E+06	7.3E+06	
Apr	7.5E+06	5.1E+06	8.1E+06	7.2E+06	1.1E+07	1.3E+07	6.7E+06	5.8E+06	6.5E+06	4.7E+06	6.0E+06	6.9E+06	6.3E+06	6.0E+06	9.1E+06	4.6E+06	5.5E+06	9.1E+06	1.0E+07	
May	6.0E+06	5.0E+06	6.2E+06	5.7E+06	1.0E+07	8.2E+06	6.3E+06	5.0E+06	5.4E+06	4.1E+06	6.3E+06	5.4E+06	5.6E+06	4.5E+06	5.7E+06	3.7E+06	8.6E+06	6.4E+06	8.3E+06	
Jun	4.8E+06	3.7E+06	5.5E+06	5.0E+06	5.7E+06	5.1E+06	5.0E+06	3.8E+06	3.9E+06	3.4E+06	4.3E+06	4.2E+06	4.2E+06	3.4E+06	4.0E+06	3.3E+06	7.2E+06	4.5E+06	5.8E+06	
Jul	3.5E+06	3.2E+06	4.6E+06	3.5E+06	4.3E+06	4.3E+06	4.9E+06	3.4E+06	3.6E+06	3.0E+06	3.3E+06	5.3E+06	4.7E+06	3.5E+06	3.2E+06	2.8E+06	1.4E+07	3.5E+06	4.4E+06	
Aug	3.2E+06	3.9E+06	3.5E+06	3.4E+06	3.4E+06	3.3E+06	4.0E+06	2.8E+06	3.2E+06	2.7E+06	2.8E+06	4.8E+06	4.5E+06	4.4E+06	3.4E+06	3.0E+06	7.7E+06	3.1E+06	4.2E+06	
Sep	2.7E+06	2.9E+06	3.4E+06	4.6E+06	3.7E+06	2.9E+06	3.6E+06	2.4E+06	3.0E+06	2.4E+06	2.5E+06	3.8E+06	5.4E+06	3.2E+06	2.9E+06	2.6E+06	5.6E+06	2.7E+06	3.2E+06	
Oct	1.8E+06	1.8E+06	2.9E+06	2.5E+06	3.5E+06	2.6E+06	3.3E+06	1.4E+06	3.0E+06	2.1E+06	2.1E+06	2.0E+06	3.0E+06	1.8E+06	2.1E+06	1.4E+06	5.2E+06	2.1E+06	2.5E+06	
Nov	3.2E+06	4.4E+06	4.5E+06	2.5E+06	7.4E+06	1.9E+06	9.0E+06	2.5E+06	2.6E+06	3.7E+06	3.1E+06	3.4E+06	4.6E+06	4.3E+06	3.0E+06	1.8E+06	7.2E+06	2.6E+06	4.3E+06	
Dec	3.4E+06	6.6E+06	5.3E+06	5.2E+06	9.2E+06	2.7E+06	8.5E+06	4.2E+06	2.8E+06	4.0E+06	5.8E+06	6.0E+06	4.6E+06	8.0E+06	2.9E+06	2.5E+06	8.8E+06	5.3E+06	3.6E+06	
Annual Values	4.4E+06	3.9E+06	5.2E+06	4.7E+06	6.1E+06	5.5E+06	5.4E+06	4.2E+06	4.2E+06	3.2E+06	3.8E+06	5.2E+06	4.9E+06	4.6E+06	4.4E+06	3.1E+06	6.4E+06	5.1E+06	6.2E+06	**4.8E+06**
(b)	**PIDDLE**																			
	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	
Jan	2.1E+06	9.1E+05	4.4E+06	3.9E+06	2.8E+06	4.1E+06	1.1E+06	4.0E+06	2.2E+06	1.2E+06	1.1E+06	3.6E+06	2.9E+06	1.8E+06	2.5E+06	1.7E+06	9.2E+05	5.0E+06	6.6E+07	
Feb	2.2E+06	1.1E+06	1.9E+06	2.2E+06	1.6E+06	3.5E+06	2.4E+06	2.3E+06	2.4E+06	9.9E+05	9.3E+05	2.5E+06	2.2E+06	2.9E+06	2.2E+06	1.5E+06	7.7E+05	3.9E+06	5.8E+06	
Mar	2.0E+06	1.9E+06	1.6E+06	1.4E+06	1.6E+06	3.1E+06	2.2E+06	1.8E+06	1.6E+06	9.4E+05	1.2E+06	3.4E+06	1.7E+06	2.2E+06	2.2E+06	1.3E+06	7.9E+05	2.7E+06	4.1E+06	
Apr	2.9E+06	2.0E+06	3.3E+06	2.5E+06	3.9E+07	5.9E+07	2.8E+06	2.4E+06	2.6E+06	1.8E+06	2.4E+06	2.8E+06	2.6E+06	2.6E+06	3.9E+06	1.8E+06	1.9E+06	4.2E+06	4.5E+07	
May	2.1E+06	1.5E+06	2.7E+06	2.2E+06	3.9E+07	3.3E+06	2.4E+06	1.9E+06	2.1E+06	1.5E+06	2.3E+06	2.1E+06	2.0E+06	1.7E+06	2.3E+06	1.3E+06	3.1E+06	2.7E+06	3.5E+06	
Jun	1.7E+06	1.3E+06	2.1E+06	1.8E+06	2.2E+06	1.9E+06	1.9E+06	1.5E+06	1.4E+06	1.2E+06	1.6E+06	1.6E+06	1.6E+06	1.2E+06	1.4E+06	1.1E+06	2.3E+06	1.8E+06	2.3E+06	
Jul	1.3E+06	1.0E+06	1.6E+06	1.3E+06	1.6E+06	1.6E+06	1.8E+06	1.2E+06	1.2E+06	8.9E+05	1.2E+06	1.8E+06	1.5E+06	1.0E+06	1.1E+06	9.0E+05	5.0E+07	1.3E+06	1.6E+06	
Aug	1.0E+06	1.3E+06	1.2E+06	1.1E+06	1.2E+06	1.2E+06	1.4E+06	9.5E+05	1.0E+06	7.9E+05	9.3E+06	1.7E+06	1.4E+06	1.2E+06	1.0E+06	9.6E+05	3.0E+06	1.1E+06	1.4E+06	
Sep	8.5E+05	1.1E+06	1.1E+06	1.2E+06	1.2E+06	1.1E+06	1.2E+06	8.5E+05	9.7E+05	7.0E+05	8.4E+06	1.4E+06	1.6E+06	8.9E+05	8.6E+05	8.6E+05	2.3E+06	9.2E+05	1.1E+06	
Oct	4.7E+05	6.2E+05	8.5E+05	7.3E+05	1.2E+06	9.3E+05	8.9E+05	4.3E+05	9.4E+05	5.2E+05	5.7E+06	7.1E+06	9.1E+05	5.0E+05	6.2E+05	4.2E+05	2.1E+06	5.7E+05	7.8E+05	
Nov	9.6E+05	1.5E+06	1.5E+06	7.9E+05	3.2E+06	7.8E+05	2.9E+06	6.9E+05	9.7E+05	9.8E+05	9.6E+06	1.1E+06	1.4E+06	1.2E+06	9.0E+05	5.6E+05	2.9E+06	7.5E+05	1.5E+06	
Dec	1.2E+06	2.6E+06	1.9E+06	1.8E+06	4.4E+06	9.8E+05	3.6E+06	1.3E+06	1.0E+06	1.3E+06	2.5E+06	2.3E+06	1.6E+06	3.3E+06	1.0E+06	7.5E+05	4.3E+06	2.4E+06	1.5E+06	
Annual Values	1.6E+06	1.4E+06	2.0E+06	1.8E+06	2.4E+06	2.4E+06	2.1E+06	1.6E+06	1.5E+06	1.1E+06	1.4E+06	2.1E+06	1.8E+06	1.7E+06	1.7E+06	1.1E+06	2.4+06	2.3E+06	2.9E+06	**1.9E+06**
Catchment										**6.6E+06**										

APPENDIX B

Appendix B presents the data produced from the CROPWAT 8.0 model (FAO 2012a). This model was used for the computation of the components of the Water Footprints of the actor 'Agriculture' in sub-section 6.3.3. The CLIMWAT 2.0 data from two stations (Exeter and Bournemouth) are presented in **Tables B1 and B2**, followed by the CROPWAT 8.0 results for the crops cultivated in the areas of Exeter (**Tables B3 i-vi**) and Bournemouth **(Table B4 i-vi**).

Table B1: Climatological Data abstracted from CLIMWAT 2.0 (FAO 2012b) for the Exeter station.

Month	Min Temp	Max Temp	Humidity	Wind	Sun	Rad	ETo	Rain	Eff rain
	°C	°C	%	km/day	hours	MJ/m²/day	mm/day	mm	mm
January	-5.7	12.4	86	432	1.7	3.1	1.2	97	81.9
February	-3.6	12.6	82	432	2.2	5.1	1.41	73	64.5
March	-3.4	15	79	441	3.6	9	2.04	64	57.4
April	-1.3	18.4	80	423	5.1	13.7	2.74	52	47.7
May	1	21.4	81	380	6.1	17.2	3.38	58	52.6
June	4.6	24.9	79	354	6.7	19	4.08	50	46
July	6.7	26.3	80	354	6.3	17.9	4.08	43	40
August	5.8	25.2	82	337	5.7	15.2	3.5	56	51
September	3.2	23	83	346	4.6	11	2.75	60	54.2
October	0.1	19.3	86	346	3	6.4	1.83	77	67.5
November	-3.1	15.4	85	380	8.8	7.4	1.26	74	65.2
December	-4.6	13.5	86	415	1.7	2.6	1.2	90	77
Average	0	18.9	82	387	4.6	10.6	2.46	794	705.2

Table B2: Climatological Data abstracted from CLIMWAT 2.0 (FAO 2012b) for the Bournemouth station.

Month	Min Temp	Max Temp	Humidity	Wind	Sun	Rad	ETo	Rain	Eff rain
	°C	°C	%	km/day	hours	MJ/m²/day	mm/day	mm	mm
January	1	7.6	85	406	1.9	3.2	0.69	89	76.3
February	1	7.8	81	406	2.8	5.5	0.97	61	55
March	2	10	78	415	3.9	9.2	1.48	66	59
April	3.5	12.8	79	389	5.7	14.4	2.04	48	44.3
May	6.5	16.1	79	372	6.8	18.1	2.72	55	50.2
June	9.5	19.3	79	346	7.2	19.6	3.26	54	49.3
July	11.3	21.4	79	337	7.1	18.9	3.45	40	37.4
August	11.1	21.1	80	337	6.5	16.1	3.03	56	51
September	9.1	18.6	84	328	5.1	11.4	2.04	66	59
October	6.9	15.1	83	346	3.5	6.8	1.38	80	69.8
November	3.2	10.8	84	372	2.5	3.9	0.88	84	72.7
December	1.7	8.6	85	397	1.8	2.6	0.68	90	77
Average	5.6	14.1	81	371	4.6	10.8	1.89	789	701.2

Tables B3: Results of the CROPWAT 8.0 model (FAO 2012a) for the crops cultivated in the Exeter area.

(i): Barley. Planting date: 10th of March.

Month	Decade	Stage	Kc	ETc	ETc	Eff rain	Irr. Req.
			coeff	**mm/day**	**mm/dec**	**mm/dec**	**mm/dec**
Mar	1	Init	0.3	0.55	0.5	2	0.5
Mar	2	Init	0.3	0.61	6.1	19.2	0
Mar	3	Deve	0.4	0.91	10	18.1	0
Apr	1	Deve	0.79	1.97	19.7	16.5	3.2
Apr	2	Mid	1.17	3.19	31.9	15.2	16.7
Apr	3	Mid	1.28	3.77	37.7	16	21.7
May	1	Mid	1.28	4.04	40.4	17.3	23.1
May	2	Mid	1.28	4.32	43.2	18	25.1
May	3	Mid	1.28	4.61	50.7	17.1	33.6
Jun	1	Late	1.26	4.83	48.3	16.1	32.2
Jun	2	Late	0.99	4.02	40.2	15.3	24.8
Jun	3	Late	0.64	2.62	26.2	14.7	11.6
Jul	1	Late	0.35	1.44	10.1	9.5	0

(ii): Maize. Planting date: 1st of April.

Month	Decade	Stage	Kc	ETc	ETc	Eff rain	Irr. Req.
			coeff	**mm/day**	**mm/dec**	**mm/dec**	**mm/dec**
Apr	1	Init	0.3	0.75	7.5	16.5	0
Apr	2	Init	0.3	0.82	8.2	15.2	0
Apr	3	Deve	0.46	1.37	13.7	16	0
May	1	Deve	0.76	2.41	24.1	17.3	6.8
May	2	Deve	1.06	3.58	35.8	18	17.7
May	3	Mid	1.31	4.75	52.2	17.1	35.1
Jun	1	Mid	1.34	5.15	51.5	16.1	35.5
Jun	2	Mid	1.34	5.46	54.6	15.3	39.3
Jun	3	Mid	1.34	5.47	54.7	14.7	40
Jul	1	Late	1.27	5.19	51.9	13.5	38.4
Jul	2	Late	0.96	3.92	39.2	12.6	26.6
Jul	3	Late	0.61	2.39	26.3	14.1	12.2
Aug	1	Late	0.38	1.42	4.2	4.8	0

(iii): Spring Wheat. Planting date: 10th of March.

Month	Decade	Stage	Kc	ETc	ETc	Eff rain	Irr. Req.	Etgreen	Etblue
			coeff	**mm/day**	**mm/dec**	**mm/dec**	**mm/dec**	**mm/period**	**mm/period**
Mar	1	Init	0.3	0.55	0.5	2	0.5	0	0.5
Mar	2	Init	0.3	0.61	6.1	19.2	0	6.1	0
Mar	3	Init	0.3	0.68	7.5	18.1	0	7.5	0
Apr	1	Deve	0.31	0.78	7.8	16.5	0	7.8	0
Apr	2	Deve	0.54	1.49	14.9	15.2	0	14.9	0
Apr	3	Deve	0.87	2.56	25.6	16	9.6	16	9.6
May	1	Mid	1.18	3.73	37.3	17.3	20	17.3	20
May	2	Mid	1.27	4.29	42.9	18	24.9	18	24.9
May	3	Mid	1.27	4.59	50.5	17.1	33.3	17.2	33.3
Jun	1	Mid	1.27	4.88	48.8	16.1	32.8	16	32.8
Jun	2	Late	1.25	5.1	51	15.3	35.6	15.4	35.6
Jun	3	Late	0.99	4.06	40.6	14.7	25.9	14.7	25.9
Jul	1	Late	0.67	2.74	27.4	13.5	13.9	13.5	13.9
Jul	2	Late	0.4	1.62	11.3	8.8	0	11.3	0

(iv): Spring Oilseed rape. Planting date: 10th of April.

Month	Decade	Stage	Kc	ETc	ETc	Eff rain	Irr. Req.
			coeff	mm/day	mm/dec	mm/dec	mm/dec
Apr	1	Init	0.7	1.75	17.5	16.5	1
Apr	2	Init	0.7	1.92	19.2	15.2	3.9
Apr	3	Deve	0.78	2.3	23	16	7
May	1	Deve	0.92	2.92	29.2	17.3	11.9
May	2	Deve	1.07	3.61	36.1	18	18
May	3	Mid	1.13	4.09	45	17.1	27.8
Jun	1	Mid	1.13	4.35	43.5	16.1	27.4
Jun	2	Late	1.13	4.61	46.1	15.3	30.8
Jun	3	Late	1.09	4.43	44.3	14.7	29.7
Jul	1	Late	1.04	4.24	16.9	5.4	10.2

(v): Winter Wheat. Planting date: 10th of October.

Month	Decade	Stage	Kc	ETc	ETc	Eff rain	Irr. Req.
			coeff	mm/day	mm/dec	mm/dec	mm/dec
Oct	1	Init	0.4	0.85	0.9	2.1	0.9
Oct	2	Init	0.4	0.73	7.3	23.2	0
Oct	3	Init	0.4	0.66	7.2	22.7	0
Nov	1	Init	0.4	0.58	5.8	21.6	0
Nov	2	Init	0.4	0.51	5.1	21.1	0
Nov	3	Init	0.4	0.5	5	22.6	0
Dec	1	Init	0.4	0.49	4.9	24.6	0
Dec	2	Init	0.4	0.48	4.8	26	0
Dec	3	Init	0.4	0.48	5.3	26.4	0
Jan	1	Init	0.4	0.48	4.8	27.5	0
Jan	2	Init	0.4	0.48	4.8	28.4	0
Jan	3	Init	0.4	0.51	5.6	26.1	0
Feb	1	Init	0.4	0.54	5.4	23.1	0
Feb	2	Init	0.4	0.56	5.6	21	0
Feb	3	Init	0.4	0.65	5.2	20.4	0
Mar	1	Init	0.4	0.73	7.3	20	0
Mar	2	Deve	0.4	0.82	8.2	19.2	0
Mar	3	Deve	0.49	1.12	12.3	18.1	0
Apr	1	Deve	0.61	1.53	15.3	16.5	0
Apr	2	Deve	0.73	1.99	19.9	15.2	4.6
Apr	3	Deve	0.84	2.48	24.8	16	8.8
May	1	Deve	0.95	3.02	30.2	17.3	12.9
May	2	Deve	1.07	3.62	36.2	18	18.1
May	3	Deve	1.19	4.3	47.3	17.1	30.2
Jun	1	Mid	1.26	4.84	48.4	16.1	32.3
Jun	2	Mid	1.26	5.13	51.3	15.3	36
Jun	3	Mid	1.26	5.13	51.3	14.7	36.7
Jul	1	Mid	1.26	5.14	51.4	13.5	37.8
Jul	2	Mid	1.26	5.14	51.4	12.6	38.8
Jul	3	Mid	1.26	4.89	53.8	14.1	39.8
Aug	1	Mid	1.26	4.65	46.5	16	30.5
Aug	2	Late	1.2	4.19	41.9	17.3	24.6
Aug	3	Late	0.81	2.65	29.1	17.6	11.5
Sep	1	Late	0.41	1.23	11.1	15.7	0

(vi): Winter Oilseed rape. Planting date: 10th of August.

Month	Decade	Stage	Kc coeff	ETc mm/day	ETc mm/dec	Eff rain mm/dec	Irr. Req. mm/dec
Aug	1	Init	0.7	2.59	2.6	1.6	2.6
Aug	2	Init	0.7	2.45	24.5	17.3	7.2
Aug	3	Deve	0.7	2.29	25.2	17.6	7.6
Sep	1	Deve	0.81	2.42	24.2	17.4	6.8
Sep	2	Deve	0.95	2.6	26	17.6	8.4
Sep	3	Mid	1.09	2.65	26.5	19.3	7.3
Oct	1	Mid	1.13	2.4	24	21.5	2.6
Oct	2	Mid	1.13	2.06	20.6	23.2	0
Oct	3	Late	1.12	1.84	20.3	22.7	0
Nov	1	Late	1.07	1.56	15.6	21.6	0
Nov	2	Late	1.03	1.31	2.6	4.2	2.6

Tables B4: Results of the CROPWAT 8.0 model (FAO 2012a) for the crops cultivated in the Bournemouth area.

(i): Barley. Planting date: 10th of March.

Month	Decade	Stage	Kc coeff	ETc mm/day	ETc mm/dec	Eff rain mm/dec	Irr. Req. mm/dec
Mar	1	Init	0.3	0.39	0.4	2	0.4
Mar	2	Init	0.3	0.45	4.5	20.5	0
Mar	3	Deve	0.39	0.65	7.2	18.6	0
Apr	1	Deve	0.75	1.39	13.9	15.8	0
Apr	2	Mid	1.09	2.23	22.3	13.8	8.5
Apr	3	Mid	1.19	2.71	27.1	14.8	12.3
May	1	Mid	1.19	2.98	29.8	16.3	13.5
May	2	Mid	1.19	3.25	32.5	17	15.4
May	3	Mid	1.19	3.46	38.1	16.8	21.2
Jun	1	Late	1.17	3.62	36.2	16.9	19.3
Jun	2	Late	0.93	3.02	30.2	16.9	13.2
Jun	3	Late	0.61	2.03	20.3	15.5	4.8
Jul	1	Late	0.34	1.16	8.2	9.1	0

(ii): Maize. Planting date: 1st of April.

Month	Decade	Stage	Kc coeff	ETc mm/day	ETc mm/dec	Eff rain mm/dec	Irr. Req. mm/dec
Apr	1	Init	0.3	0.56	5.6	15.8	0
Apr	2	Init	0.3	0.61	6.1	13.8	0
Apr	3	Deve	0.45	1.02	10.2	14.8	0
May	1	Deve	0.72	1.79	17.9	16.3	1.6
May	2	Deve	0.99	2.69	26.9	17	9.8
May	3	Mid	1.22	3.53	38.9	16.8	22.1
Jun	1	Mid	1.24	3.83	38.3	16.9	21.4
Jun	2	Mid	1.24	4.05	40.5	16.9	23.6
Jun	3	Mid	1.24	4.13	41.3	15.5	25.8
Jul	1	Late	1.18	3.99	39.9	13	26.9
Jul	2	Late	0.9	3.1	31	11.3	19.7
Jul	3	Late	0.59	1.94	21.4	13.2	8.2
Aug	1	Late	0.38	1.2	3.6	4.7	0

(iii): Spring Wheat. Planting date: 10^{th} of March.

Month	Decade	Stage	Kc	ETc	ETc	Eff rain	Irr. Req.
			coeff	mm/day	mm/dec	mm/dec	mm/dec
Mar	1	Init	0.3	0.39	0.4	2	0.4
Mar	2	Init	0.3	0.45	4.5	20.5	0
Mar	3	Init	0.3	0.5	5.5	18.6	0
Apr	1	Deve	0.31	0.57	5.7	15.8	0
Apr	2	Deve	0.52	1.07	10.7	13.8	0
Apr	3	Deve	0.82	1.86	18.6	14.8	3.8
May	1	Mid	1.11	2.76	27.6	16.3	11.3
May	2	Mid	1.19	3.24	32.4	17	15.3
May	3	Mid	1.19	3.45	37.9	16.8	21.1
Jun	1	Mid	1.19	3.66	36.6	16.9	19.7
Jun	2	Late	1.17	3.82	38.2	16.9	21.2
Jun	3	Late	0.94	3.11	31.1	15.5	15.7
Jul	1	Late	0.64	2.17	21.7	13	8.6
Jul	2	Late	0.39	1.34	9.4	7.9	0

(iv): Spring Oilseed rape. Planting date: 10^{th} of April.

Month	Decade	Stage	Kc	ETc	ETc	Eff rain	Irr. Req.
			coeff	mm/day	mm/dec	mm/dec	mm/dec
Apr	1	Init	0.7	1.3	13	15.8	0
Apr	2	Init	0.7	1.43	14.3	13.8	0.5
Apr	3	Deve	0.77	1.74	17.4	14.8	2.7
May	1	Deve	0.89	2.23	22.3	16.3	6
May	2	Deve	1.02	2.77	27.7	17	10.7
May	3	Mid	1.08	3.12	34.3	16.8	17.5
Jun	1	Mid	1.08	3.31	33.1	16.9	16.3
Jun	2	Late	1.08	3.5	35	16.9	18.1
Jun	3	Late	1.03	3.43	34.3	15.5	18.8
Jul	1	Late	0.98	3.33	13.3	5.2	6.8

(v): Winter Wheat. Planting date: 10th of October.

Month	Decade	Stage	Kc	ETc	ETc	Eff rain	Irr. Req.
			coeff	mm/day	mm/dec	mm/dec	mm/dec
Oct	1	Init	0.4	0.64	0.6	2.2	0.6
Oct	2	Init	0.4	0.55	5.5	23.6	0
Oct	3	Init	0.4	0.49	5.3	23.8	0
Nov	1	Init	0.4	0.42	4.2	23.9	0
Nov	2	Init	0.4	0.35	3.5	24.2	0
Nov	3	Init	0.4	0.33	3.3	24.7	0
Dec	1	Init	0.4	0.3	3	25.3	0
Dec	2	Init	0.4	0.27	2.7	25.9	0
Dec	3	Init	0.4	0.27	3	25.7	0
Jan	1	Init	0.4	0.27	2.7	26.2	0
Jan	2	Init	0.4	0.27	2.7	26.4	0
Jan	3	Init	0.4	0.31	3.4	23.7	0
Feb	1	Init	0.4	0.35	3.5	19.9	0
Feb	2	Init	0.4	0.39	3.9	17.2	0
Feb	3	Init	0.4	0.46	3.7	18	0
Mar	1	Init	0.4	0.53	5.3	19.8	0
Mar	2	Deve	0.4	0.6	6	20.5	0
Mar	3	Deve	0.48	0.81	8.9	18.6	0
Apr	1	Deve	0.59	1.1	11	15.8	0
Apr	2	Deve	0.7	1.43	14.3	13.8	0.4
Apr	3	Deve	0.8	1.82	18.2	14.8	3.4
May	1	Deve	0.91	2.26	22.6	16.3	6.3
May	2	Deve	1.01	2.75	27.5	17	10.5
May	3	Deve	1.12	3.25	35.7	16.8	18.9
Jun	1	Mid	1.18	3.64	36.4	16.9	19.6
Jun	2	Mid	1.18	3.85	38.5	16.9	21.6
Jun	3	Mid	1.18	3.93	39.3	15.5	23.8
Jul	1	Mid	1.18	4	40	13	27
Jul	2	Mid	1.18	4.08	40.8	11.3	29.4
Jul	3	Mid	1.18	3.91	43	13.2	29.8
Aug	1	Mid	1.18	3.74	37.4	15.6	21.8
Aug	2	Late	1.13	3.41	34.1	17.2	16.9
Aug	3	Late	0.77	2.08	22.9	18	4.9
Sep	1	Late	0.4	0.95	8.5	16.8	0

(vi): Winter Oilseed rape. Planting date: 10th of August.

Month	Decade	Stage	Kc	ETc	ETc	Eff rain	Irr. Req.
			coeff	mm/day	mm/dec	mm/dec	mm/dec
Aug	1	Init	0.7	2.21	2.2	1.6	2.2
Aug	2	Init	0.7	2.12	21.2	17.2	4
Aug	3	Deve	0.7	1.9	20.9	18	2.8
Sep	1	Deve	0.79	1.87	18.7	18.7	0
Sep	2	Deve	0.91	1.86	18.6	19.6	0
Sep	3	Mid	1.03	1.88	18.8	20.8	0
Oct	1	Mid	1.06	1.71	17.1	22.3	0
Oct	2	Mid	1.06	1.47	14.7	23.6	0
Oct	3	Late	1.06	1.29	14.2	23.8	0
Nov	1	Late	1.01	1.06	10.6	23.9	0
Nov	2	Late	0.97	0.85	1.7	4.8	1.7